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Co-benefits of Sustainable Forestry

Ecological Studies of a Certified Bornean Rain Forest

Springer

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Front cover: A Borneo elephant in a reduced-impact logged forest in the Deramakot Forest Reserve in Malaysia (photo by Andreas Langner). *Back cover: Left*: A tropical rain forest after conventional selective logging (photo by Andreas Langner). *Middle*: A pristine tropical rain forest in Deramakot (photo by Andreas Langner). *Right*: A tropical rain forest abutting converted land (photo by Masaaki Takyu)

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Preface

This volume is a compilation of the ecological studies that colleagues and I have conducted as a Malaysia-Japan collaborative project in the Deramakot Forest Reserve, Sabah, Malaysia. The Deramakot Forest Reserve is a logging concession that is managed directly by the Sabah State Government. What makes Deramakot outstanding is its ecologically benign improved logging system. Deramakot is a model site of sustainable forest management with reduced-impact logging, which promotes natural regeneration of commercial trees.

"Sustainable forest management" has been an ambiguous and confusing term for many of us because the original concept, dating back to many decades ago, was developed for producing timber near permanently, but practices on the ground failed in most parts of the tropics, leading to extensive tropical deforestation. Many people are indeed skeptical about the effectiveness of sustainable forest management. Hard-core conservationists often think that it is an ill-advised concept in terms of its negative influences on the conservation of ecosystems and biota. Apart from adequacy of the concept of "sustainable forest management," we are charged to develop a true synergy system, which can integrate forestry and biodiversity conservation in tropical rain forests. It is practically impossible to convert all tropical forests to a strictly protected park amid increasing demands for land. We need to harvest timber, fibers, and other resources from certain forests to support human well-being on a local scale as well as in the global context. Failure of management resulting from the failure of society will eventually lead to a collapse of the whole ecosystem. The most pragmatic approach will be the synergy of forestry and conservation. We need to revisit the concept of sustainable forest management in the midst of the challenges of global changes and land conversions.

In the latest (tenth) meeting of the Conference of the Parties (COP10; Nagoya 2010) of the Convention on Biological Diversity (CBD), ambitious conservation goals of the Aichi Biodiversity Targets were officially adopted. One of the targets is the conservation of 17 % of terrestrial and inland water areas of the world by 2020. The concept of sustainable forest management will be increasingly important to meet the Aichi Targets while demands for land are also increasing. Recently, I attended several international expert meetings relating to CBD. I faced obstinate

opposition by hard conservationists against adopting "sustainable forest management" as a tool to accomplish conservation goals. Such strict conservationists often ignore the important role of forestry to provide local people with subsistence. This realization prompted me to publish the results of our ecological studies as a book so that our research results can be available in many parts of the world and all stakeholders can carefully examine the ecological co-benefits of the sustainable forest management system that is implemented in Deramakot.

We spent nearly 7 years conducting the ecological studies that are described in this book. All researchers in this project are ecologists with pure academic interests and backgrounds. Therefore, I believe that the research results reflect our neutral positions. This book is written primarily to convey our scientific findings. Nevertheless, I as an editor would like to convey an important message underlying this book: that is, that legitimate governance and capacity combined with careful planning based on ecological principles are needed to accomplish the synergy of forestry and conservation in natural tropical rain forests.

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We are particularly grateful to Datuk Sam Mannan, Director of Sabah Forestry Department, who has shepherded Deramakot and is a strong advocate of ecologically benign sustainable forest management; he intends to duplicate this system in all other management units in Sabah. We are grateful also to Mr. Ewin Rawinder Ajon and the staff members of the Deramakot Forestry Office, who helped us in all aspects. Prof. Tohru Nakashizuka (formerly at Research Institute of Humanity and Nature) assisted us in the early phase of our project. We dedicate this volume to those hard-working foresters and scientists who have invented and started and are implementing the sustainable forest management project in Deramakot. Without their efforts, it would not have been possible for us to conduct our research project and to convey our important messages to related stakeholders of the world.

Kyoto, Japan Kanehiro Kitayama

Color Plates



Plate 1 A remnant stand of the pristine lowland tropical rain forest in Deramakot. This stand is now protected as a conservation area. Dominant canopy trees largely consist of dipterocarp trees. (Photograph by Andreas Langner)



Plate 2 A moderately degraded, logged-over lowland tropical rain forest in Deramakot. The elapsed time after the last logging is unknown. (Photograph by Andreas Langner)



Plate 3 A conventionally logged, lowland tropical rain forest in Sabah. Collateral damage to the ground is obvious. (Photograph by Andreas Langner)



Plate 4 A conventionally logged, lowland tropical rain forest entangled with massive climbing bamboos in Tangkulap. Tangles of climbing bamboos prevent the regeneration of canopy trees. (Photograph by Masaaki Takyu)



Plate 5 A log yard in one of the compartments of Deramakot with a remnant forest stand in the background. (Photograph by Kanehiro Kitayama)



Plate 6 A group of Borneo elephants roaming in a logged-over lowland tropical rain forest in Deramakot. (Photograph by Hisashi Matsubayashi)

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Chapter 1
Management History of the Study Sites: The Deramakot and Tangkulap Forest Reserves

Robert C. Ong, Andreas Langner, Nobuo Imai, and Kanehiro Kitayama

1.1 Introduction

The Deramakot Forest Reserve and the Tangkulap Forest Reserve, where the ecological studies described in this book were conducted, are located in the interior of the state of Sabah, Malaysia (north Borneo) (Fig. 1.1). In Sabah, forested lands, including both natural and planted forests, are found mainly within areas designated by law as Forest Reserve, State Park, or Wildlife Sanctuary. Together these three classes of forest lands account for 53% (3.87 million ha) of Sabah's landmass, and are collectively referred to as the Permanent Forest Estate (PFE). Little native forest is currently left outside the boundaries of the PFE. Forest Reserves are managed by the Sabah Forestry Department and a large part of these are designated by law primarily to produce timber from natural forests. Despite declining timber output, forestry remains a dominant land use in Sabah.

The major source of timber supplies in Sabah comes from selective harvesting in natural forests. Generally speaking, production forestry with selective logging has been extremely destructive to the forest environment in Sabah, largely because of inappropriate harvesting practices, as well as the relatively rich timber stands found in Sabah. Excessive damage caused by logging retards the process of forest recovery and regeneration and, as a consequence, undermines the long-term sustainability of the forest for timber production (Pinard et al. 2000). The sustainable management of forest resources is a matter of serious concern, and several initiatives have been undertaken by the Forestry Department during the past 20 years to improve forest management in Sabah.

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Fig. 1.1 Location of Sabah, Malaysia

A key initiative towards improving forest management began with the development of a model forest management plan for one Forest Management Unit (FMU), the Deramakot Forest Reserve. Sabah's entire Forest Reserve (Fig. 1.2) is divided into 23 FMUs. A Forest Management Unit (FMU) is the basic entity for management planning. The main idea was to adopt this model management planning approach for the other FMUs. In 1995, a model management plan for the Deramakot Forest Reserve was completed under the auspices of the Malaysian German Sustainable Forest Management Project.

The 23 FMUs are sequentially numbered. The mean area of the 23 FMUs is approximately 100,000 ha each. The Deramakot Forest Reserve corresponds to FMU 19A and has been managed directly by the Sabah Forestry Department since 1989. The Tangkulap Forest Reserve, corresponding to FMU 17A, was managed by a private logging company before 2002 and transferred to the direct management of the Sabah Forestry Department after 2002. Twelve FMUs have been licensed out to various private companies under a 100-year concession agreement as of 2010; the remaining FMUs still remain to be licensed for concession agreement or are directly managed by the Sabah Forestry Department.

This chapter mainly describes the management of Deramakot over the past 15 years in comparison to Tangkulap. The two FMUs are located side by side, but they have contrasting logging histories. The Deramakot Forest Reserve has been

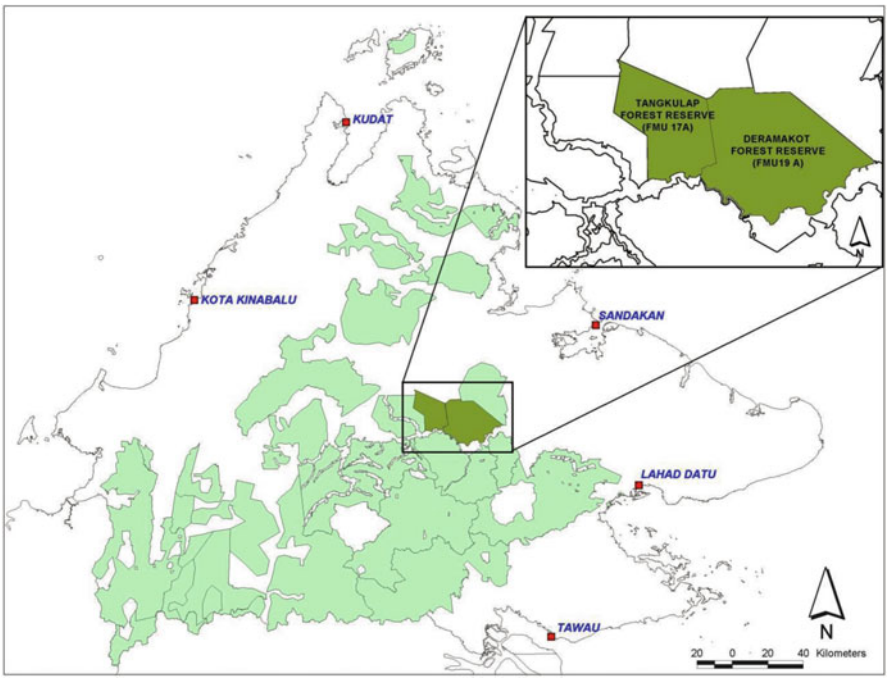


Fig. 1.2 Map of Sabah showing the location of the Deramakot Forest Reserve and the Tangkulap Forest Reserve, and the extent of the Class II Production Forest in the state of Sabah. Class II forest, which is divided into 23 FMUs, is specially designated for commercial timber production. Class II forest other than Deramakot and Tangkulap is indicated with *light green*. *FMU*, forest management unit

sustainably managed by improved logging practices since 1989, whereas the Tangkulap Forest Reserve has been managed by destructive "conventional" logging practices. Ecological comparisons are made between these two forest reserves throughout this book. As is shown, the pre-logging original vegetation and other physiological environments are similar with each other between the two forest reserves. The research approach throughout this book is a comparative approach to test the main effects of logging practices on ecosystem structure and function and on biodiversity.

1.2 Natural Setting of the Deramakot and Tangkulap Forest Reserves

Located in central Sabah, both the Deramakot and Tangkulap Forest Reserves are legally designated under the Sabah Forest Enactment as Class II Production Forests, meaning that these units are primarily intended for commercial timber production.

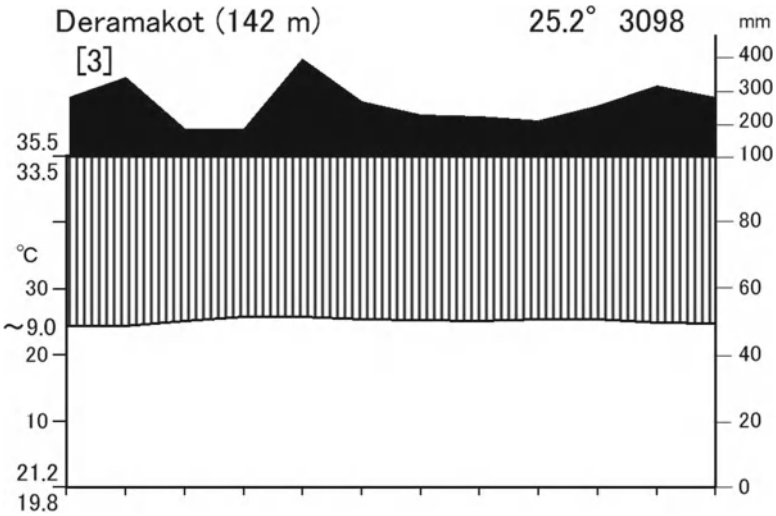


Fig. 1.3 Climate diagram of Deramakot, Sabah, Malaysia

In comparison, forests designated as Class I Protection Forests are so designated to protect environmentally sensitive areas from major human disturbances, such as logging. Deramakot covers an area of approximately 55,083 ha; Tangkulap, at 27,550 ha, is much smaller (Fig. 1.2).

The mean daily temperature is 25.2°C, with a mean daily minimum of 19.8°C–21.2°C and a daily maximum of 33.5°C–35.5°C at the Deramakot field operations complex (142 m above sea level) located in the interior of the Deramakot Forest Reserve (Fig. 1.3). Mean annual rainfall for the period 2008–2010 recorded at the same complex is 3,098 mm. Although monthly rainfall exceeds 100 mm for any month (except for El Niño drought years), March and April are relatively dry while November and December are relatively wet. Therefore, monthly rainfall exceeds potential evapotranspiration for any month, representing a typical tropical rain forest climate. However, droughts can severely influence the tropical rain forests of this region during El Niño, which occurs at irregular intervals (generally 5–10 years). The mortality of standing trees increased from 0.8 to 1.1% in non-El Nino years to 2.5–2.6% during the 1998 El Nino (Aiba and Kitayama 2002), which was one of the greatest droughts in the past three decades.

The natural forests in the two reserves are largely composed of logged-over mixed lowland dipterocarp forests. Mixed lowland dipterocarp forests are one of tropical rain forests in equatorial Southeast Asia and once occurred on well-drained soils in the entire interior area of Sabah (and elsewhere in equatorial Southeast Asia) below 600 m above sea level, but are now confined to the PFE because of past land-use and forestry activities. A primary mixed lowland dipterocarp forest is dominated by the family Dipterocarpaceae (which includes the genera *Dipterocarpus*, *Shorea*, *Parashorea*, *Hopea*, *Dryobalanopsis*, and *Vatica*) and consists of highly diverse assemblages of canopy tree species in the range of 200–250 species per

Table 1.1 Absolute and relative areas of six different slope classes for each of the Tangkulap and Deramakot Forest Reserves

	Tangkulap		Deramakot	
Slope classes	Area (ha)	Percentage (%)	Area (ha)	Percentage (%)
0–5°	22,967	82	20,060	36
5–10°	4,350	16	19,979	36
10–15°	498	2	11,814	21
15–20°	114	<1	3,488	6
20–25°	23	<1	357	1
>25°	1	<1	60	<1

hectare. The average canopy height exceeds 60 m. The above-ground biomass of primary forest is quite high. A recent meta-analysis demonstrated that the average above-ground biomass of primary forests across Borneo is 457 Mg ha^{-1}, about 60% higher than the Amazonian average of 289 Mg ha^{-1} (Slik et al. 2010). Accordingly, mixed lowland dipterocarp forests are considered as the most species-rich and gigantic tropical rain forests among the three major tropical regions of the world.

There are several types of mixed lowland dipterocarp forests in equatorial Southeast Asia. The one in the Deramakot Forest Reserve and the Tangkulap Forest Reserve is the most widespread type, dominated by multiple red seraya (section Brachypterae, Mutica, Ovalis, and Pachycarpae of *Shorea*), yellow seraya (section Richetioides of *Shorea*), and selangan batu (section Shorea of *Shorea*). Large-statured dipterocarp species, with slender trunks and buttress roots, often exceed a main canopy stratum and form a discontinuous emergent stratum. Several other species also, such as *Syzygium* spp. (Myrtaceae), *Palaquium* spp., *Madhuca* spp. (Sapotaceae), *Heritiera* spp., and *Scaphium* spp. (Sterculiaceae), are major constituent species of a main canopy stratum. Several woody lianas, such as *Spatholobus* spp. (Leguminosae), *Strychnos* spp. (Loganiaceae), and *Willughbeia* spp. (Apocynaceae) often climb to the canopy stratum. On the other hand, forests degraded by past logging activities have a smaller number of dipterocarp species and are dominated by pioneer tree species such as *Macaranga* spp. (Euphorbiacea), *Croton* spp. (Euphorbiacea), *Neonauclea* spp. (Rubiaceae), *Neolamarckia cadamba* (Rubiaceae), and *Octomeles sumatrana* (Datiscaceae) and climbing bamboos *Dinochloa* spp. (Poaceae) (see Chap. 3 for detailed floristic composition).

The landscape of Deramakot is largely made up of gently undulating terrain and low hills. More than 64% of the area occurs on slopes steeper than 5° (Table 1.1). Deramakot has an altitudinal range of 12–355 m a.s.l., with a mean of 135 m. Generally, Tangkulap is less hilly than Deramakot (Fig. 1.4). The majority of the area (82% of the area) occurs on flatter slopes, less than 5° (Table 1.1). The altitude in Tangkulap ranges from 30 to about 300 m a.s.l., with a mean of 83 m.

Approximately 91% of the soils in Deramakot are classified as Lokan Association, whereas the remaining is of the Dalit Association (Acres et al. 1975). Both soil associations are derived from mudstone and sandstone parent materials, where

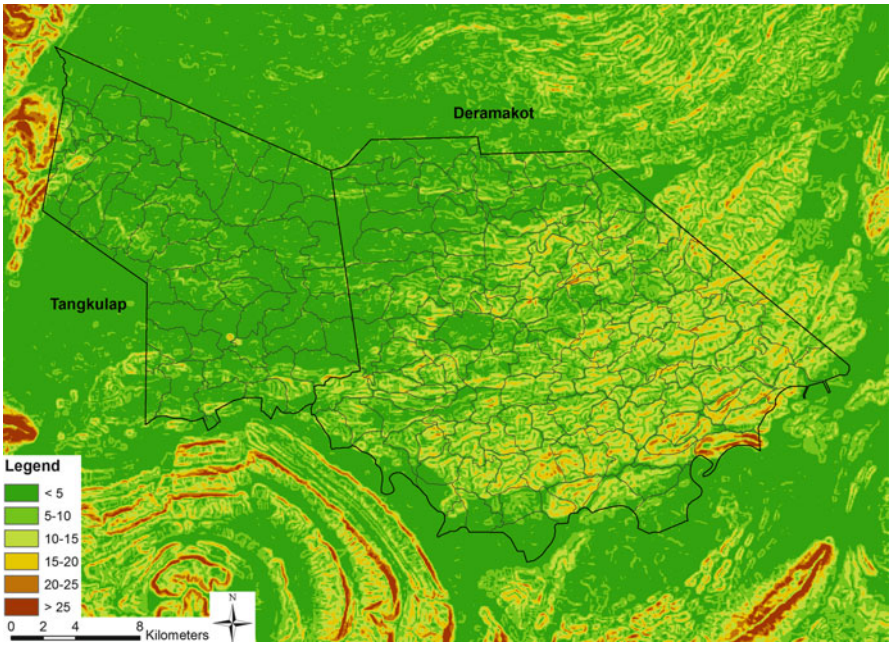


Fig. 1.4 Slope classification for the Tangkulap and Deramakot Forest Reserve [values in degrees (°)]

orthic acrisols are the dominant soils. Lokan and Dalit are rated as marginal in terms of suitability for agricultural development. Similar to Deramakot, the predominant soil association in Tangkulap is the Lokan Association, covering about 94% of the reserve, with the Dalit Association comprising the remaining 6%.

1.3 Management and Logging History

1.3.1 Deramakot: Logging Before 1989

Prior to 1989, logging was conducted by licensed, private concessionaires. Commercial logging commenced in the Deramakot Forest Reserve in 1956, but not until 1961, when the chainsaw was first introduced and tractor use became more common, did timber production levels exceed 10,000 m^3 per year. Annual timber production was 74,065 m^3 in 1961 and peaked at 467,757 m^3 in 1973. Based on available production figures in relationship to area worked, the average production from selective logging in the primary lowland forest of Deramakot for the period 1962–1968 was about 110 m^3 ha^{-1}. In pristine forest logging, this translates to about 12 trees ha^{-1} >60 cm dbh (diameter at breast height) during a period when logging was highly selective. Between 1975 and 1985, much of Deramakot was logged a

second time. The exact size and location of these areas cannot be determined from available records. The last logging license issued in Deramakot was in 1987. At about the same time, approximately 4,500 ha in the northern part of Deramakot was reported to have been logged illegally. Apart from the prescription of a minimum dbh cutting limit, logging in Deramakot before 1989 was not closely controlled or supervised by the Forestry Department; that is, logging contractors were left on their own to plan logging operations. Between 1962 and 1984, refinement treatments involving the cutting of climbers and the poisoning of defective and noncommercial trees were carried out over an area of about 21,000 ha of Deramakot.

1.3.2 Deramakot After 1989: Introduction of Sustainable Forest Management

During this period, Deramakot was directly managed by the State Government (i.e., the Sabah Forestry Department). In 1989, under the auspices of the Malaysian German Sustainable Forest Management Project, the Sabah Forestry Department ceased all logging activities in Deramakot. In 1992, the Sabah Forestry Department began work toward formulating a forest management plan for the Deramakot Forest Reserve. The ultimate aim was to produce a model management plan that conformed to internationally acceptable standards and principles for sustainable forest management (Kleine and Heuveldop 1993). The Deramakot forest management plan was completed in 1995, providing management direction and specific targets over a planning period of 10 years, that is, 1995–2004 (Sabah Forestry Department 1995). Logging with the improved logging practices of reduced-impact logging (RIL) resumed in Deramakot following the forest management plan in 1995.

A forest management plan outlines the long-term management goals, specific targets, prescribed activities, and their implementation schedule over a specified planning period. In other words, the management plan serves as an "enabling environment" in which the systematic implementation and monitoring of all management prescriptions may be carried out and followed through. Compliance with the management plan and any standing guidelines are closely monitored and assessed against an annual work plan.

As defined in the forest management plan, Deramakot is designated primarily for sustainable timber production from the natural forest. About 89.6% (or 49,374 ha) of Deramakot is zoned for timber production and about 7% (3,605 ha) of the area is zoned for protection (Fig. 1.5). The production zone refers to areas designated for timber production and where forest regeneration is by natural regeneration (assisted or otherwise) or enrichment planting. The protection zone refers to areas excluded from harvesting largely because of steepness of terrain. For operational planning purposes, Deramakot is divided into 135 compartments ranging in size from 110 to 900 ha. Compartment boundaries are delineated according to natural features such as ridges and rivers, as well as roads. Because virtually all of Deramakot has been logged over, it has an extensive network of logging roads. Including the main road, secondary roads, and feeder roads, Deramakot has a road density of 3.9 m ha^{-1}.

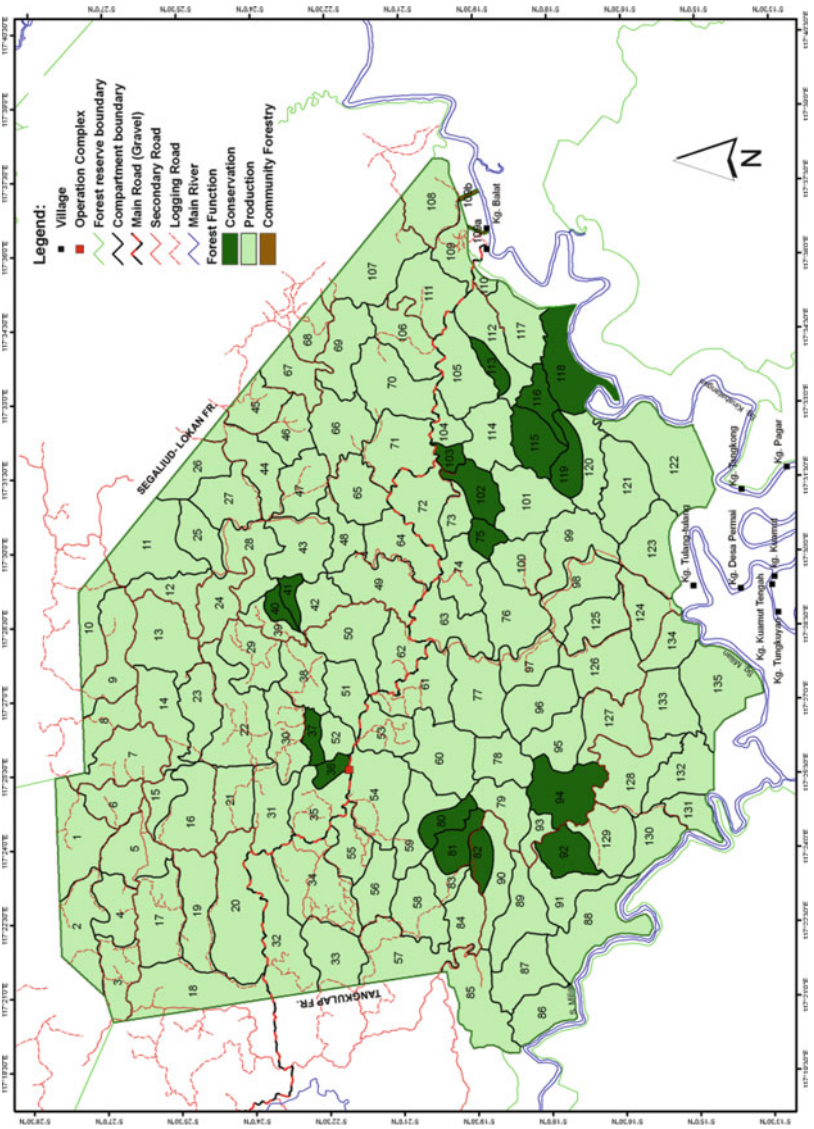


Fig. 1.5 Deramakot Forest Reserve showing the delineation of 135 compartments and their zoning for either conservation or production. Compartments designated as conservation areas are indicated with *dark green* and are excluded from timber harvesting. Two to three compartments are currently harvested annually

The volume of annual harvested logs has been consistently less than 15,000 m^3 since logging resumed according to the new forest management plan in 1995. Logging is implemented on a compartment basis in rotation. The number of compartments to be logged is generally fewer than four per year to allow regrowth in unlogged compartments, which translates to a 40-year rotation cycle for all Deramakot. A compartment will only be harvested when commercial timber stand exceeds 25 m^3/ha, to maintain forest structure and to avoid invasion of pioneer species.

In 1997, the management of the Deramakot Forest Reserve was certified as having fulfilled the requirements of a "well-managed" forest according to the principles and criteria set by the Forest Stewardship Council (FSC). It was the first and only natural forest area in the tropics to be certified at that time according to internationally acceptable standards. For Deramakot, continued compliance with the management plan is done by an independent assessor as part of its certification requirement.

1.3.3 Tangkulap Before 2002

Logging was licensed out to private concessionaires during this period in Tangkulap. Commercial timber exploitation in Tangkulap started in the 1970s and ceased around the mid-1980s. Second-entry logging began in 1995 and continued until year 2002. During this period, about 3,000 ha of pristine forests in Tangkulap was also logged. Historical records of the volume of commercial timber extracted from Tangkulap are not complete, and therefore it is difficult to ascertain the amount of timber extracted from the area. Poor logging practices have resulted in overlogging and a forest extremely poor in commercial regeneration. Overall, the forest of Tangkulap is of poorer quality than that of Deramakot as of 2002 through 2010 because of the highly intensive and destructive logging practices, particularly during the second-entry logging period.

The day-to-day management of Tangkulap was transferred to the Sabah Forest Department in 2002. All logging activities have been banned since then to allow the recovery of timber resources. The ecological studies described in the following chapters were conducted during the period 2003–2010, that is, the period after logging ceased and the forest was a heterogeneous mosaic of poorly stocked timber stands in various stages of recovery.

1.3.4 Quantification of Logging Disturbances During the Period Before Our Studies

It is critical to compare logging histories and disturbances quantitatively between the two forest reserves during the past few decades to explain present-day differences of forest ecosystems. However, historical records of the volume of harvested

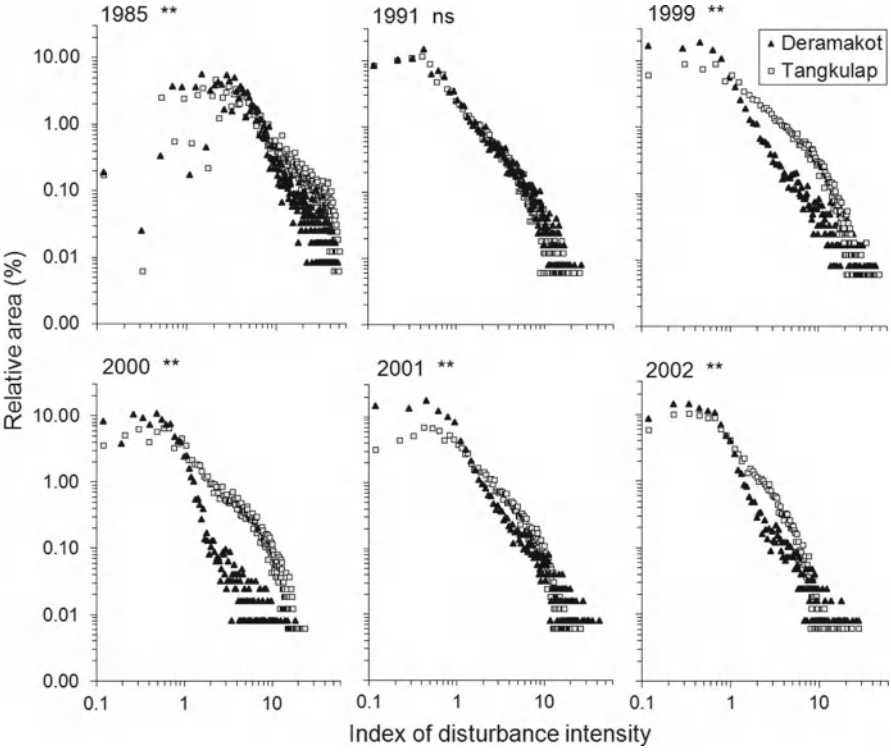


Fig. 1.6 Intensity of logging disturbances and relative areas affected by logging on double logarithmic scales from 1985 and 2002. Intensity of logging disturbances and relative areas were estimated using archive Landsat satellite data in Deramakot (*filled triangles*) and Tangkulap (*open squares*). Greater disturbances are derived from greater reflectance values of the red spectrum, most likely reflecting the occurrence of bare soils caused by logging activities (see text for details). *Convex curve* indicates occurrence of greater intensities of disturbances in larger areas

timber and the localities where logging took place are missing in the Tangkulap Forest Reserve, where conventional logging practices persisted until 2002. We analyzed the sitings of logging by making use of archived remote sensing data in the two reserves between 1985 and 2002 (Fig. 1.6).

The algorithm of this analysis is based on stronger reflectance values of the red band on bare soils compared with intact vegetation. The red spectrum is absorbed by green leaves for photosynthesis, while it is reflected on bare soils. High reflectance values of the red band are thought to represent two types of disturbances in this area: one is related to selective logging in forest stands (e.g., creation of skid trails, and denudation caused by uprooted trees and tractors), and the other is related to permanent infrastructures such as major logging roads. Effects of the former type of disturbances are ephemeral whereas those of the latter are long lasting. Reflectance values of the red band can combine both effects and become a snapshot of disturbances. Natural landslides can give rise to a false signal of disturbance, but they are rare in

both reserves. We analyzed Landsat MSS, TM, and ETM+ scenes of the study area over a period of 18 years from 1985 until 2002. All Landsat images were resampled to 80-m pixel size to unify the spatial resolution. Clouds and cloud shadows were masked and removed to avoid false results because the spectral signal of the red band was influenced by atmospheric effects. However, the effects of haze could not be completely removed by masking clouds and cloud shadows. To remove the noise from haze, we applied a 3×3 kernel moving window approach and extracted abrupt changes of red reflectance only within each window. The background idea for this approach is that haze shows gradual smooth changes over larger areas and that it does not influence the relative relationships of the reflectance values among the neighboring pixels within a window. The slopes of the reflectance values of the red band in each kernel window were derived as disturbance intensity. Percent relative area was plotted on the *y*-axis against disturbance intensity on the *x*-axis double logarithmically in Fig. 1.6.

The curves were more convex in Tangkulap than in Deramakot except for 1991, suggesting that much heavier logging occurred in greater areas in Tangkulap at these periods. It is highly likely that these convex curves correspond to unregulated conventional logging in Tangkulap. The curve in Tangkulap became steeper, reflecting the governmental suspension of logging activities after 2001. The curve of Deramakot was consistently steeper than the curve of Tangkulap after 1999 in spite of timber harvest (RIL), suggesting either restricted sites of logging or reduced impacts per site resulting from improved logging practices, or both effects. Greater expanses of high red reflectance were still clearly visible in Tangkulap in 2002.

1.4 The Forest of Deramakot and Tangkulap Today

Current land-cover types of the Tangkulap and Deramakot region were mapped with the help of the Geographic Information System (Fig. 1.7). Altogether, 22 different land-cover types could be distinguished in this region. Table 1.2 shows the absolute and relative areas of each detected land-cover type for Tangkulap and Deramakot, respectively. The most striking differences between the two forest reserves can be seen in the relative areas of closed, high-density evergreen lowland forests and the two bare soil classes. The relative area of undisturbed dipterocarp forests is much greater in the Deramakot Forest Reserve than in the Tangkulap Forest Reserve. The area of bare soil or bare soil with fragmented vegetation is negligible also in Deramakot. In contrast to Deramakot, about 8% of Tangkulap is covered with both bare soil classes, which clearly reflects denser logging roads used to extract timber resources in Tangkulap. The percentage of heavily degraded lowland forests including forest regrowth (evergreen lowland regrowth in Table 1.2) is 10% in Tangkulap, whereas it is only 4% in Deramakot, highlighting the higher impact on the forests by recent and former logging events in Tangkulap.

The land-cover map for Deramakot (Fig. 1.7) shows a tendency for the greater expanse of poor forest to be located in the north and south whereas the better forest

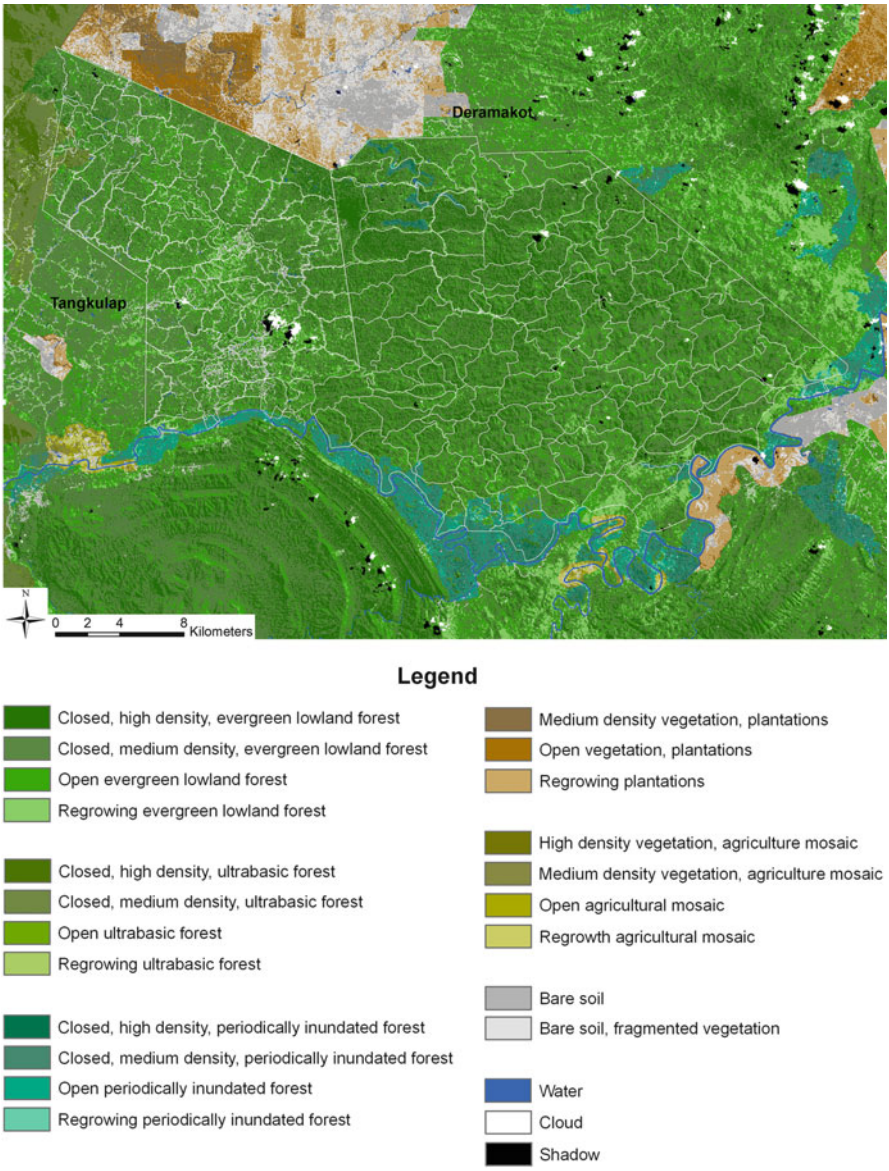


Fig. 1.7 Land-cover classification of the study area. *Light grey outlines* indicate compartment units of the two forest reserves

stretches across the central region from east to west, particularly in the west-central region. Harvesting records show that logging in Deramakot started from the south and southeastern part of Deramakot along the Kinabatangan River and progressed

Table 1.2 Absolute and relative extents of land-cover types in each of the Tangkulap and Deramakot Forest Reserves

Forest type	Tangkulap		Deramakot	
Closed, high-density, evergreen lowland	3,001 ha	(11%)	13,145 ha	(24%)
Closed, medium-density, evergreen lowland	13,382 ha	(49%)	26,173 ha	(48%)
Open evergreen lowland	5,831 ha	(21%)	10,463 ha	(19%)
Evergreen lowland regrowth	2,655 ha	(10%)	2,470 ha	(4%)
Closed, high-density, ultramafic	65 ha	(<1%)	–	–
Closed, medium-density, ultramafic	264 ha	(1%)	–	–
Open ultramafic	30 ha	(<1%)	–	–
Ultramafic regrowth	18 ha	(<1%)	–	–
Closed, high-density, riverine	3 ha	(<1%)	324 ha	(1%)
Closed, medium-density, riverine	27 ha	(<1%)	1,400 ha	(3%)
Open riverine	27 ha	(<1%)	665 ha	(1%)
Riverine regrowth	4 ha	(<1%)	112 ha	(<1%)
Bare soil, fragmented vegetation	1,255 ha	(5%)	94 ha	(<1%)
Bare soil	727 ha	(3%)	35 ha	(<1%)
Water	85 ha	(<1%)	19 ha	(<1%)

upward toward the north. The second entry for logging also followed a similar pattern, but as access from the north improved, concurrent second-entry logging operations also started from the north. Although a large part of Deramakot has been logged at least twice since the first logging in 1956, records show that the west-central region of Deramakot was logged only once (between 1974 and 1978). This information provides a plausible explanation for the higher concentration of moderate and good forests in the west-central region. The better-quality forests also appear to be associated with hilly terrain. One likely explanation for the proximate cause of the poor forest in the northern portion of Deramakot is the relatively flat terrain. Such topography makes it easy for tractors to operate, which may well explain the currently poor timber stands in such areas. In addition, about 4,500 ha in the northern part of Deramakot was reported to have been illegally logged sometime between 1986 and 1988.

1.5 Current Silvicultural Prescriptions and Management in Deramakot

Silvicultural operations in Deramakot under the present management may be divided into three main activities: (1) harvesting, (2) timber stand improvement, and (3) enrichment planting. All these operations are prescribed by compartment and carried out by private companies under the close supervision of the Forestry Department.

1.5.1 Harvesting

Timber production has been the main focus of management for Deramakot. Harvesting is carried out generally using crawler tractors following RIL guidelines (Lohuji et al. 2009). RIL basically refers to a set of harvesting guidelines that are designed to reduce logging damage. The guidelines include specifications for harvest planning and operations using ground-based skidders, namely, the tractor (Pinard and Putz 1996). A key feature of RIL is the requirement of a comprehensive harvest plan before the commencement of harvesting operations (Fig. 1.8). This plan includes a predetermined layout of feeder roads and skid trails that facilitates the systematic and orderly extraction of timber while significantly reducing soil disturbance. Skid trails must be designed to avoid riparian reserves and stream buffers. Steep areas above 25° must be avoided also. Moreover, canopy opening after logging must not be more than 0.1-ha gap size.

Harvesting in mixed dipterocarp forest is selective in terms of tree size and species. Under RIL guidelines, the minimum cutting limit is generally maintained at 60 cm dbh, and a maximum cutting limit is imposed at 120 cm dbh. All trees that bear large fruits as a food source for wildlife (such as mangos and durians) must not be harvested for timber. Tree felling must follow a directional felling approach so as to reduce damage to the environment and potential crop trees. Rainy days must be avoided for harvesting to protect topsoil. Harvesting is conducted by a contractor and closely supervised for enforcement and control by forestry officers who are stationed at the district forest office inside Deramakot. Implementation of RIL must strictly follow RIL guidelines (obtainable at http://www.deramakot.sabah.gov.my/PDF/RILOperationGuideBook.pdf).

The annual allowable cut for the first planning period (1995–2005) was set at 15,000 m^3; over the second planning period (2005–2014), the annual allowable cut is set at 18,000 m^3. Overall, however, the prescribed harvest levels have never been consistently achieved throughout the planning period (Table 1.3). In the early years, this was attributed to administrative constraints pertaining to harvesting contracts, but in recent years wet weather has been the main cause of low log extraction rates because logging operations are prohibited during rainy days. Over time, as the forest recovers, the sustainable annual allowable cut will likely increase.

All logs produced from Deramakot are sold by auction. A study of timber yield from second-entry logging for 19 compartments logged between 1995 and 2006 found the average yield from the net logged area to be about 55 $m^3\ ha^{-1}$, corresponding to 8.6 trees ha^{-1} (Gobilik et al. 2010). The dipterocarps account for 90% of the total timber output, with keruing (*Dipterocarpus* spp., 28.7%) and red seraya (*Shorea* spp., 23.2%) making up the largest proportion of yield. To reduce waste wood, the salvaging of residual timber was introduced in 2003, which was also to meet the requirement of the sustainability principles of the FSC. Salvaged timber constitutes those parts of the tree not extracted during the main timber extraction operations, such as short logs (short stems), hollow logs, lower stem (butt end) parts, and large branches.

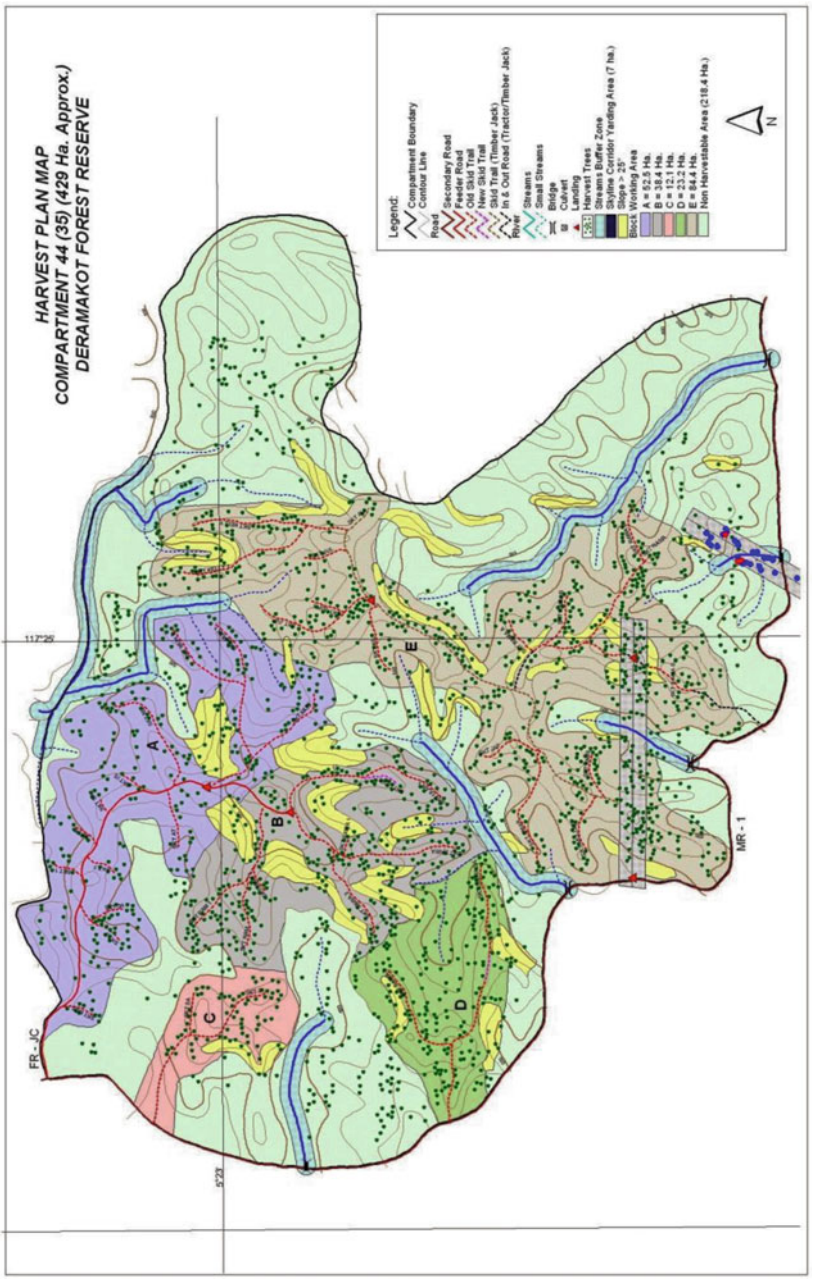


Fig. 1.8 Example of a comprehensive harvest plan. Shown is the comprehensive harvest plan for Compartment 44 (429 ha) of the Deramakot Forest Reserve. The locations of harvestable trees are indicated with *green dots*, with predetermined layout of feeder roads (*red lines*) and skid trails (*red dotted lines*)

Table 1.3 Deramakot log production from 1997 to 2009

Year	Log production (m^3)	Salvaged residual logs (m^3)	Total (m^3)
1997	15,909	–	15,909
1998	11,857	–	11,857
1999	1,051	–	1,051
2000	13,409	–	13,409
2001	15,591	–	15,591
2002	17,211	–	17,211
2003	14,555	822	15,377
2004	19,822	1,808	21,631
2005	11,425	484	11,909
2006	16,129	1,869	17,998
2007	13,662	705	14,637
2008	6,357	1,376	7,733
2009	11,450	943	12,393

1.5.2 Enrichment Planting

The management plan specifies a target of 200 ha of enrichment planting to be carried out annually in areas that have been determined to be poorly regenerated. Enrichment planting involves the planting of commercial indigenous species, mainly dipterocarps, in clusters or as single trees along corridors cut through the forest. The number of trees planted per hectare ranges from 180 to 300 trees, depending on forest conditions. Plantings are tended three times a year for the first 3 years. From 1996 to 2009, 1,370 ha of enrichment plantings was established in Deramakot.

1.5.3 Timber Stand Improvement

This treatment refers to the cutting of climbing bamboo, woody vines, and the selective removal of defective and noncommercial trees to promote the growth of seedlings and pole-size trees (usually not more than 25 cm dbh) of commercially desirable species. In Deramakot, under the present management, the cutting of climbing bamboo and woody vines is prescribed as a blanket treatment in compartments designated for timber production. However, figs (which are an important food source for birds and wildlife) are not cut at all, and woody vines are not cut within stream buffers (30 m on either side of a permanent stream). During the current management planning period (2005–2014), a specified target of 2,000 ha is to be treated annually. In a period of 15 years, that is, since 1996 to 2010, some 25,000 ha in Deramakot has been treated.

The superficial appearance of logged-over forests, with few large commercial trees, may suggest lack of regeneration. However, moderately degraded forest stands do support the regeneration of young commercial trees in terms of seedlings/saplings, as is shown later (see Chap. 3). The successful establishment of such abundant seedlings/saplings can be arrested if climbing bamboos and woody vines persist over them.

Therefore, heavily shaded seedlings/saplings need to be liberated from heavy vine loads or sometimes from non-commercial trees. From a silvicultural perspective, the adequacy of regeneration is usually defined as having a sufficient number of commercial trees (below the minimum cutting limit) to yield an economic harvest within a cutting cycle of 30–40 years (Gan and Weinland 1995). Any yield above 40 m^3 per hectare is generally considered sufficient for an economic cut. In a second cut where trees are expected to be just above the minimum cutting limit (60 cm dbh), this would translate to about seven or eight trees per hectare.

Although the density of commercial regeneration was found to be generally adequate, the proportion of defective stems in Deramakot is considered high overall (Ong and Kleine 1995). The high proportion of defective trees is cause for concern because it could invariably undermine the potential of the forest to produce a future crop. The proportion of defective trees for those trees of 5–25 cm dbh was found to be higher in the logged forests (51%) than in the old growth forest (32%). This condition is likely the result of damage caused during previous logging and the detrimental effects of vines that proliferate and persist after logging. Therefore, the removal of vines to secure the survival and quality of potential crop trees is important in silviculture treatment.

Both climbing bamboo and woody vine densities are high in the logged forests of Deramakot. More than 80% of all trees ≥5 cm dbh are found to be affected by vines (Ong and Kleine 1995). The difference in densities between logged forest and old growth forest was particularly obvious for climbing bamboo, indicating that climbing bamboo is perhaps more invasive than are woody vines and therefore presents a more serious silvicultural problem. The problem of vines in forest regeneration is not only confined to the dipterocarp forests of Southeast Asia. Vidal et al. (1997) reported from a study in eastern Amazonia that vine density was three times greater in young building-phase forest than in mature forest. The detrimental effects of vines on trees are a widely recognized problem in the regeneration of logged forests in the tropics (Putz 1991). The benefits of such timber stand improvement practices have been reviewed by Wadsworth (1997). Cutting vines and bamboos to liberate seedlings and saplings is often criticized as having negative consequences for biodiversity protection. However, Edwards et al. (2010) suggest that such intensive silvicultural treatments are beneficial to restore the biodiversity of degraded forests by allowing faster regrowth of remaining trees. Moreover, cutting of woody vines before logging is a commonly recommended silvicultural practice aimed at reducing felling damage (Appanah and Putz 1984; Fox 1967).

Leaving logged forests with their high proportion of pioneers, defective trees, and vine tangles untreated allows suboptimal growing conditions to persist, thereby limiting the future wood production capacity of the forest as well as the rate of succession. Furthermore, because seeding in logged forests with few mother trees is uncertain and unreliable, every effort to secure the survival of regeneration and to promote their growth by silvicultural treatments would be beneficial. Compartment-level inventories carried out for the entire production area of Deramakot indicate that about 75% of Deramakot may be naturally regenerated through the judicious application of timber stand improvement treatments. The remaining areas, however, will need some form of artificial regeneration, such as enrichment planting or tree plantations.

The logged forests of Deramakot and Tangkulap support a relatively high proportion of pioneer trees (see Chap. 3). *Macaranga* spp. are the most common pioneer trees found in the logged forest of Deramakot. *Macaranga* is widespread throughout Sabah, and their dominance is a typical characteristic of logged dipterocarp forests (Primack and Lee 1991; Whitmore 1984). *Macaranga* is characterized by rapid early growth, and, being extremely light demanding, they are quick to colonize exposed soils in large gaps such as those created by logging. In timber stand improvement, any attempt to remove *Macaranga* is not recommended until their growth and development have stabilized and canopy closure has been attained, that is, usually not earlier than 5 years after logging (Weidelt 1993).

1.6 Forest Certification

As has been stated, the improved forest management in the Deramakot Forest Reserve has been certified according to the standards set by FSC since 1997. With the growing concern over deforestation and its associated environmental effects, an increasing number of consumers of forest-based products such as timber wish to be assured that the products that they purchase are produced through sustainable forest management practices. In this regard, forest certification has been promoted as a market-linked mechanism to encourage sustainable forest management practices. The question of long-term sustainability has become a major issue surrounding forest management today, and the global nature of forestry has been borne on tropical countries to improve forest management practices. To be certified, forest management practices are evaluated against a set of principles and criteria for good forest management. Therefore, certification ensures greater accountability and transparency in management.

The management of Deramakot has benefited in many ways from the process of certification. For instance, since the certification of Deramakot as a "well-managed" forest in 1997, logs produced from Deramakot have commanded significantly higher prices than those produced from other uncertified sources (Kollert and Lagan 2005). In addition, certification ensures incremental improvement of all aspects of management in Deramakot.

1.6.1 Consideration for Biodiversity Conservation

Although forests managed for timber may not be a perfect way for conserving biodiversity, logged forests can serve to preserve various forms and high levels of biodiversity, as the following chapters of this book demonstrate. One of the criteria of FSC certification requires the monitoring and safeguarding measures of biodiversity. The Sabah Forest Department has been conducting, to fulfill the criterion, a regular aerial census of wild orangutans, which are the flagship species of Bornean tropical rain forests (Ancrenaz et al. 2005). A census conducted in 2000 estimates the population of orangutans in Deramakot at about 900, roughly 1.7 individuals/km^2. The aerial census so far suggests that logged forests in Deramakot

still have an important role in orangutan conservation. Therefore, there appears to be common ground between the management of natural forests for timber and for wildlife conservation.

1.6.2 Consideration for Local Communities

One of the criteria of the FSC also requires special considerations for local communities. Although there are no communities present within Deramakot, it is the policy of the Sabah Forestry Department to encourage the participation of local communities in forest management activities wherever possible. This policy is also consistent with the requirements of all existing forest certification schemes including the FSC. A special committee was set up to address issues and opportunities for village communities living in close proximity to Deramakot and Tangkulap. Meetings with representatives of these villages are conducted two to three times a year.

Management activities such as timber stand improvement, planting, and boundary demarcation largely employ a local workforce. For instance, in 2008, the Department engaged members of a local community from the nearby village of Balat to carry out forest restoration over a 100-ha patch of degraded forest in the southern part of Deramakot. In addition to providing livelihood opportunities, the Sabah Forestry Department also continues to allow gravity-fed water to be piped from Deramakot to two nearby villages.

Topic Outlook of Production Forestry in Sabah

By Robert C. Ong

Conceptually, the Government of Sabah has put in place a sound system of forest management for production forests, but implementation of the system is constrained by poorly stocked logged-over forests. Funding the restoration of these forests is a central issue of forestry in Sabah today. Should funds be made available, timber-related activities in natural forests would best focus on timber stand improvement and restoration. For purely economic reasons, it appears more than likely that short-rotation forest plantations will eventually replace natural forests as the principal source of commercial timber in Sabah. This being the case, some focus in research should be given to explore ways of making plantations more eco-friendly. Given sufficient time, natural forests can still be expected to contribute to timber production as their timber productivity is gradually restored. However, production from natural forests will probably be best managed for high-value timber crops on long cutting cycles of not less than 35 years, and any timber stand improvement activities in logged forests would best be considered as a restorative measure rather than an economic venture. For timber production, the model of forest management most likely to

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succeed in the future would be a combination of intensively cultivated forest plantations and extensively managed natural forests. In addition, provisions for environmental services, as well as recreational use, are expected to become a more prominent aspect of integrated forest management.

The socioeconomic challenges confronting forestry pose a more formidable challenge to forest management than technical issues such as silviculture. Amidst mounting pressure for arable land, a key challenge to sustainable forest management in Sabah and elsewhere is to lobby for public support, as well as to convince policy makers, to maintain and protect the tenure of lands specifically designated for forestry use, for example, forest reserves, and to recognize that this particular class of land as an asset will remain more valuable under forest cover than as other forms of land use. Therefore, there must be a firm government resolve to implement forestry as a legitimate use of land, either by legislative decree or strict forest policies, thereby removing the option for conversion. In this regard, the importance of political support in ensuring the implementation of sustainable forest management is crucial. Toward this end, the success of Deramakot has been instrumental in promoting the idea of sustainable forest management to policy makers and political leaders alike. Because it demonstrates various aspects of good forest management in practice, Deramakot has served as an important educational tool for changing public perception of forest management in Sabah. Since gaining recognition as a well-managed forest, Deramakot has become host to regular educational visits from university students, teachers, and policy makers, as well as nongovernmental organizations (NGOs) with special interest in forestry. Such recognition has been beneficial for the overall public perception and image of forestry. Visits to Deramakot by key political leaders have given sustainable forest management in Sabah its much-needed political endorsement.

The outlook of production forestry depends on how the Sabah Government maintains the relevance of forestry in a changing socioeconomic environment. The Forestry Department has a crucial role to play in this regard. Success at this role will depend on how the Department manages the transition of forestry from a largely exploitation phase to a phase of multiple use and sustainable management. How to add socioeconomic value to logged forests, even during a prolonged cessation of timber production caused by past overuse, is the challenge confronting forestry in Sabah and elsewhere in the tropics. The importance of highly innovative and effective technical agencies, such as the Forestry Department, in surmounting such a challenge cannot be overemphasized. The ultimate test of competency for the Department (it is the department that needs to influence the policy makers in government) will include the following: influencing land-use policy and public opinion in support of forestry, sourcing for badly needed funds for forest restoration, and adapting to the changing values of a society that is becoming increasingly influential in decision making.

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Chapter 2
The Application of Satellite Remote Sensing for Classifying Forest Degradation and Deriving Above-Ground Biomass Estimates

Andreas Langner, Jupiri Titin, and Kanehiro Kitayama

2.1 Introduction

Forests are defined as areas that are covered with trees taller than 5 m in height and at least 10% canopy cover, integrating both pristine forests and anthropogenic forest plantations. In the subtropical and tropical zones, a 10–40% canopy cover defines an open canopy forest, and a 40–100% canopy cover is classified as a closed canopy forest (FAO 2000a). Pristine forest ecosystems, especially in the tropics, are severely threatened by degradation and deforestation caused by human impact, whereby forest degradation is often a precursor for deforestation (Asner et al. 2005). Forest degradation is characterized by a significant reduction in tree density from closed to open or fragmented forests (Achard et al. 2004; DeFries et al. 2007). Deforestation, on the other hand, is described as the reduction of forest cover to less than 10% (FAO 2000b; Mayaux et al. 2005).

Regarding the global carbon budget, information about forest status is very important as forests can be either sinks or sources of atmospheric carbon. Additionally, detailed knowledge about the current extent and degradation status of forest ecosystems is crucial to apply and assure sustainable forest management. There are various methods of monitoring forest ecosystems. Monitoring techniques on the ground show severe restrictions as only punctual data can be obtained, thus lacking information about the whole area (Tangki and Chappell 2008).

In contrast to ground-based monitoring techniques, the application of remote sensing in combination with ground surveys offers the possibility to directly obtain

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information over larger areas. Especially when the forest reserves to be monitored are vast, remote, and to a large extent inaccessible on the ground, remote sensing provides the most feasible way of assessing land-cover changes and deforestation rates on a regional scale (Asner et al. 2010). Depending on the sensor characteristics, remote sensing generally allows detecting various land-cover properties as well as monitoring land-cover and land-use changes through time.

The objective of this chapter is to analyze remote sensing data of the two forest reserves in Deramakot and Tangkulap, as described in Chap. 1, to estimate the effects of sustainable forest management and reduced-impact logging in contrast to conventional logging regarding forest degradation and the conservation of above-ground biomass. We examine to what extent reduced-impact logging can conserve above-ground biomass in comparison to the more destructive conventional logging, which is applied in most production forests in the tropics.

2.1.1 Forest Status with Focus on Southeast Asia

The following paragraph describes some examples of successful forest monitoring programs at various scales, such as the global land-cover database International Geosphere–Biosphere Programme, Data and Information System (IGBP-DIS; Loveland et al. 2000) or the Tropical Ecosystem Environment Observations by Satellite (TREES) project (Mayaux et al. 1998; Achard et al. 2002a, b). Results show that pressure on forests is not homogeneous throughout the world. In contrast to boreal forests, the situation of the pantropical forests is quite alarming (FAO 2000a). Tropical deforestation has increased tremendously in the past three decades (Fuller 2006). Based on the statistics from the forest resources assessment report (FAO 2000a), the tropics suffered under a net loss of natural forests of 12.3 Mha per year during 1990–2000. Tropical rain forests globally exhibited an annual loss of 5.8 ± 1.4 Mha between 1990 and 1997, and additionally an area of 2.3 ± 0.7 Mha becomes degraded every year (Achard et al. 2002a; Mayaux et al. 2005).

A detailed analysis per continent showed that the Southeast Asian humid tropics are most severely threatened. Although Latin America and Africa show an annual deforestation rate of 0.4% to 0.5%, respectively, the annual net loss of forest area in Southeast Asia increased from 0.8% between 1990 and 2000 to almost 1.0% for the period 2000–2005 (FAO 2006). Other independent studies support these results (Achard et al. 2002a; Mayaux et al. 2005). According to a study by the FAO (2006), only 33.4% of the land area of South and Southeast Asia was still covered by forest as of 2005.

Indonesia and Malaysia, the largest countries of insular Southeast Asia, experienced even higher annual change rates of 1.3% (Marcoux 2000). In a period of a half century, Indonesia lost 64 Mha of forest from its original 162 Mha in 1950 (FWI/GFW 2002). Insular Southeast Asia experienced an annual deforestation rate of 1.2% between 2000 and 2010 (Miettinen et al. 2011). In comparison to all other Indo-Malay forests, Borneo still has one of the largest tropical rain forests remaining in Southeast Asia. However, these forests are under severe pressure, which was shown by a detailed analysis of the

forest cover of Borneo, revealing an annual deforestation rate of 1.7% between 2002 and 2005 (Langner et al. 2007). According to a study by van der Werf et al. (2009), the annual emissions from land-use change through deforestation and forest degradation account for more than 12% of the total anthropogenic emissions of greenhouse gases, thus rendering deforestation and forest degradation the second largest human-induced source of greenhouse gas emissions.

2.2 Methods of Forest Monitoring in Deramakot

2.2.1 Technical Background to Satellite Remote Sensing

The principle of the remote sensing process is to detect electromagnetic energy, which is reflected or emitted from features of the earth's surface (Lillesand et al. 2004). All energy detected by satellite sensors has to pass the atmosphere at least one time. Depending on the pathlength through the atmosphere, prevailing atmospheric conditions, the magnitude of the energy signal, and the involved wavelengths, the atmosphere has varying effects on the intensity and spectral composition of radiation reaching the sensor.

Remote sensing sensors can be characterized according to their spatial, spectral, radiometric, and temporal resolution. The spatial resolution refers to the size of a pixel, which is recorded in a raster image. Typically pixels of satellite sensors may correspond to square areas ranging in side length from several decimeters to 1,000 m or more. The spectral resolution refers to the wavelength width of different frequency bands and is also related to the number of different frequency bands recorded on a satellite platform. Radiometric resolution refers to the number of different radiation intensities, which a sensor is able to distinguish. Temporal resolution is the frequency of satellite overpasses, and is relevant in time-series studies, in which information has to be derived at certain time intervals.

Sensors can be categorized to active versus passive types. Although active sensors such as radar supply their own source of energy, passive electronic-optical systems only sense naturally available energy such as various parts of sunlight or heat radiation. Both active and passive sensors show different advantages and drawbacks, and depending on the field of application an appropriate sensor system has to be carefully selected.

2.2.2 Brief Overview of Sensor Systems for Forest Monitoring

In our study, we aimed at deriving information about the forest cover with a special focus on the level of forest degradation and the amount of above-ground biomass. However, measuring forest degradation is technically more complicated than monitoring deforestation (DeFries et al. 2007). Deriving above-ground biomass

estimates is even more challenging, as the electromagnetic energy detected by a sensor does not directly indicate biomass (Saatchi et al. 2007). The signal of all available satellite sensor types tends to saturate at a lower value than the actual above-ground biomass value (Rosenqvist et al. 2003; Gibbs et al. 2007). Because tropical rain forests are among the ecosystems with largest biomass and complex structures, correctly estimating above-ground biomass in such forests is challenging primarily because of the saturation problem.

To resolve the problem of sensor saturation and to best derive information about forest degradation and above-ground biomass, airborne light detection and ranging (LiDAR) sensors have to be considered. Forest canopy, bare ground features, intermediate forest structures, and understory can be discriminated, thus best allowing the calculation of the biomass of the monitored vegetation (Asner 2009; Ballhorn et al. 2009). However, because of its narrow field of view and the high costs of data acquisition, this system does not allow a large-scale application. As a surrogate of extensive field inventory, the use of LiDAR data for calibration purposes in combination with satellite data is nevertheless promising (Asner et al. 2010).

In contrast to airborne data, spaceborne systems are able to cover much larger areas because of their wider range of view. Active radar systems with their cloud-penetrating sensing capabilities provide great advantages in comparison to optical data (Asner 2001). However, radar data do not provide as much detailed information in comparison to optical imagery to distinguish different land-cover or vegetation classes (Podest and Saatchi 2002; Fuller 2006). Further disadvantage of the application of radar data in Deramakot with its heterogeneous topography is the fact that radar backscatter is strongly affected by topographic slopes (Lillesand et al. 2004; Gibbs et al. 2007), thus rendering the interpretation of the land-cover information difficult.

Corrections for topographic effects are less problematic with passive optical sensors. Additionally, the multispectral sensing capabilities of these systems allow higher accuracies to distinguish different types of vegetation. However, one of the principal limitations of the use of optical imaging systems in the humid tropics is that frequent cloud cover hampers the detection of land-cover information (Fuller 2006). Haze also obscures land-cover monitoring based on optical sensors. Additionally, optical data also show restrictions in estimating biomass stocks of tropical forests with certainty (Thenkabail et al. 2004). Much research has been undertaken in the humid tropics using Landsat data (Steininger 2000; Foody et al. 2001, 2003; Tangki and Chappell 2008). According to Apan (1997), crown cover density provides important information to identify the successional stages of a forest. The canopy of tropical rain forests becomes more uneven with increasing age and biomass, resulting in higher rates of self-shadowing by emergent trees (Weishampel et al. 2001), which can be detected using optical data.

2.2.3 Satellite and Ground Truth Data Used During the Study

Being aware of the problem of sensor saturation at a lower biomass value (Lu 2005) as well as the heterogeneous topography of our study area, we decided to use optical

Landsat imagery for analyzing the forest degradation and the above-ground biomass of both forest reserves. A Landsat Enhanced Thematic Mapper Plus imagery (ETM+; spatial resolution of 30 m; position according to Worldwide Reference System path 117, row 56), acquired on July 9, 2000, was used as a master scene to develop an index describing the forest degradation status. Based on this index, an above-ground biomass model was developed. The year 2000 shows a combination of the impacts of different logging systems on forest degradation and above-ground biomass. Although the forests of Tangkulap were logged until 2002 using a conventional logging technique, logging has been banned in Deramakot since 1989. In 1995, reduced-impact logging was introduced to Deramakot as improved logging methodology (see Chap. 1).

However, as a result of the lack of biomass reference levels of both forest reserves, it was impossible to analyze the effects of the different forest management systems on above-ground biomass based on only the Landsat scene in 2000. To obtain the rate of annual above-ground biomass changes in each forest reserve, another ETM + scene from May 26, 2007, as well as one Landsat Thematic Mapper (TM) scene from May 22, 1991, were processed. Additionally, short- and long-term effects, such as forest degradation as well as secondary succession in forest patches where logging had been suspended, were analyzed.

Furthermore, there were also technical reasons to choose the year 2000 as the master scene to derive the biomass model. As already described, clouds and haze severely affect the quality of optical satellite data for land-cover monitoring. Fortunately, the 2000 Landsat scene shows almost no cloud contamination over both forest reserves and therefore proved optimal for the analysis. Additionally, the selected satellite scene does not depict any kind of timber stand improvements, such as vine cutting or the removal of noncommercial trees by girdling or cutting. Such activities have a great impact on the spectral properties of the monitored forests and therefore bias the analysis.

To improve the analysis of the Landsat data, the heterogeneous topographies of the two forest reserves were corrected using the digital elevation model of the Shuttle Radar Topography Mission (SRTM) dataset (90-m spatial resolution).

In addition to remote sensing data, field-based biomass measurements from 80 permanent sample plots were used to physically correlate the Landsat reflectance values to biomass measurements on the ground. The plots were sampled in 2007 for Deramakot and in 2008 for Tangkulap. Each plot consists of a circle with a radius of either 15 m or 20 m. Within these plots, all trees larger than 10 cm diameter at breast height (dbh) were measured for dbh. These values were then converted into above-ground biomass values using the allometric equation for tropical rain forests published by Brown (1997):

$$\mathrm{AGB} = \mathrm{e}^{-2.134+2.53\times\ln(\mathrm{dbh})}$$

Assuming that 49% of the measured weight is carbon, the biomass values of all sample plots were converted to carbon weight (Imai et al. 2010).

The coordinates of every permanent sample plot were obtained with a Global Positioning System (GPS). Ground truth information about the forest condition and

crown cover openness of each sample plot was collected for validation purposes, and two helicopter flights were undertaken in 2010 to derive detailed information of the crown cover condition.

2.2.4 Concept of Above-Ground Biomass Model

We developed an algorithm to estimate above-ground biomass first by referring to the degradation status of a forest. The following paragraphs give an overview of our approach to estimate the crown cover condition and forest degradation stage, which we correlated with the biomass values measured in the permanent sample plots. Subsequently, an above-ground biomass model was developed and validated, and the biomass estimates for both forest reserves were analyzed spatially and temporally. Detailed information on the single steps of data processing and the algorithms involved as well as the validation process can be found in Langner et al. (2012).

2.2.4.1 From Satellite Data to Crown Cover Condition

The basic idea behind this approach is to derive the degree of forest degradation by measuring the crown cover condition. A study by Baccini et al. (2008) demonstrated that the vegetation structure, such as crown size, tree density, texture, and shadows, best correlates with the reflectance values of the short-wave infrared bands.

According to our own empirical analysis of the different Landsat bands and band combinations, we identified the mid-infrared band (MidIR; Landsat band 7) with 2.08–2.35 μm for the Landsat TM sensor and 2.09–2.35 μm for the Landsat ETM+ sensor to best represent the crown cover status. This band is sensitive to vegetation water content but also can be used to identify bare soil (Lillesand et al. 2004). This distinction is very important, because pristine vegetation with a closed crown cover is generally wetter with a higher humidity captured within the forest than a forest with an open crown cover (Whitmore 1985; Holdsworth and Uhl 1997). Self-shadowing by single emergent trees also influences the reflectance values of that band, which can be used to identify mature forest (Adams et al. 1995; Lu et al. 2003).

There were also illumination artifacts caused by the heterogeneous topography, influencing the reflectance values as measured by the satellite. These artifacts, which bias a correct estimation of the crown cover condition, were corrected using the sun-canopy-sensor (SCS) model by Gu and Gillespie (1998). However, as the terrain of the study site shows very small scale changes in its topography, the digital elevation data with a spatial resolution of 90 m did not allow a complete removal of all illumination artifacts. Additionally, an atmospheric adjustment similar to dark object subtraction (Chavez 1988) was performed, resulting in an illumination- and atmosphere-corrected MidIR (MidIR$(b7)_{corr}$) band.

Additionally to the atmospheric adjustment, a haze correction was performed. As the blue band (Landsat band 1: 0.45–0.52 μm for TM and 0.45–0.515 μm for ETM+) is most sensitive to atmospheric effects such as haze, this band was modified [$\text{blue(b1)}_{\text{mod}}$] and used to correct the $\text{MidIR(b7)}_{\text{corr}}$ band. Although this haze correction procedure partially compensated for the effects of haze, it did not totally remove all atmospheric influences. Therefore, haze still affected the correct estimation of the crown cover condition to some extent.

Furthermore, crown cover closure can easily be detected using the MidIR band. Although bare soil is characterized by high reflectance values in the MidIR spectrum, the undisturbed crown cover of pristine forests shows much lower reflectance values. Young vegetation, regrowth, or a directly exposed herb and shrub layer, which are characteristics of logged-over forests, show intermediate reflectance values. Roughly speaking, a higher degree of logging impact results in elevated reflectance values in the MidIR band.

In our analysis, we used the reciprocal of the illumination-, atmosphere-, and haze-corrected MidIR reflectance values to derive the crown cover and forest status ($\text{CCFS}_{\text{corr}}$) index:

$$\text{CCFS}_{\text{corr}} = \frac{1}{\text{MidIR(b7)}_{\text{corr}} - 0.3 \times \text{blue(b1)}_{\text{mod}}}$$

2.2.4.2 From Crown Cover and Forest Status to Above-Ground Biomass

Before performing a regression analysis, we temporally corrected the biomass measurements of the permanent sample plots to reflect the biomass of the year 2000 when the Landsat scene was obtained. The temporal adjustment was done using an algorithm that assumes an asymptotic biomass increment of 1–2% per year (Clark et al. 2001).

We also spatially adjusted the areas of the permanent plots, because two different plot sizes were employed. For that purpose, the biomass values of both plot sizes were normalized to fit the area of one Landsat pixel with 900 m^2, corresponding to a circular plot with a 16.93-m radius. In a next step, a regression analysis between the $\text{CCFS}_{\text{corr}}$ index and the normalized and temporally adjusted above-ground biomass measurements of all permanent sample plots was performed. Because mixed lowland dipterocarp forests only were sampled, this index was calibrated to this special forest type and could not be applied to other vegetation types. Therefore, all other land-cover and land-use types, such as ultramafic forests, plantations, and agricultural areas, were masked. This mask was derived by an unsupervised ISODATA classification (100 clusters and 200 iterations; applied on the Landsat TM and ETM+ bands 1–5 and 7) of the study area.

A second-order polynomial equation was fitted and the following above-ground biomass model was developed:

$$AGB_{Landsat} = 0.009 \times CCFS_{corr}^{2}$$

Based on this model, the above-ground biomass values over the whole study area were estimated in tons C/ha. This model is sensitive to all kinds of water bodies within a forest. Their presence leads to an overestimation of the crown cover quality, thus resulting in an overestimation of the above-ground biomass values. Cloud- or haze shadows have similar effects. Therefore, an accurate land-cover classification is crucial to mask such areas. However, a complete removal of these pixels was not feasible because of slight inaccuracies of the classification result. For that reason, all areas with extraordinary high biomass values were masked in a subsequent step, assuming that the high biomass values represented such overestimations. A threshold of 645 t C/ha was empirically derived.

A random permutation test with 1,000 iterations was used to validate the AGB model based on the Landsat ETM+ scene of the year 2000 against the actually measured AGB values of the corresponding sample plots; estimated values were not significantly different from measured values at $P = 0.05$.

2.3 Results and Discussion

2.3.1 Results of the Above-Ground Biomass Estimates for the Year 2000

The above-ground biomass values for 81,778 ha over the two forest reserves were estimated (Fig. 2.1). An analysis of the results of the biomass model for the Landsat scene in 2000 demonstrated a mean above-ground biomass of 152.2 t C/ha (130.6–178.1 t C/ha) over the whole study area.

Although the mean above-ground biomass value in the year 2000 is 122.5 t C/ha (104.9–142.8 t C/ha) in Tangkulap, it is 167.9 t C/ha (143.7–195.7 t C/ha) in Deramakot. Therefore, Deramakot has a greater mean above-ground biomass by 45.3 t C/ha (38.8–52.8 tC/ha) than Tangkulap as of the year 2000. A bootstrap sampling method revealed that the biomass values between the two forest reserves were significantly different at a 95% confidence interval.

The difference of 45.3 t C/ha (38.8–52.8 tC/ha) is remarkably similar to the result of a study by Imai et al. (2009), who suggested a mean difference of 52 t C/ha between the two forest reserves as of 2001. The slightly higher estimate by Imai et al. (2009) most probably relates to the different approach of their study, using only four timber stock classes to extrapolate the biomass over the two forest reserves. However, the mean difference of 45.3 t C/ha (38.8–52.8 tC/ha) between the two forest reserves in the year 2000 can only partially explain the effects of the different management systems. Reason for this is the lack of information about a common biomass reference level before the change in management.

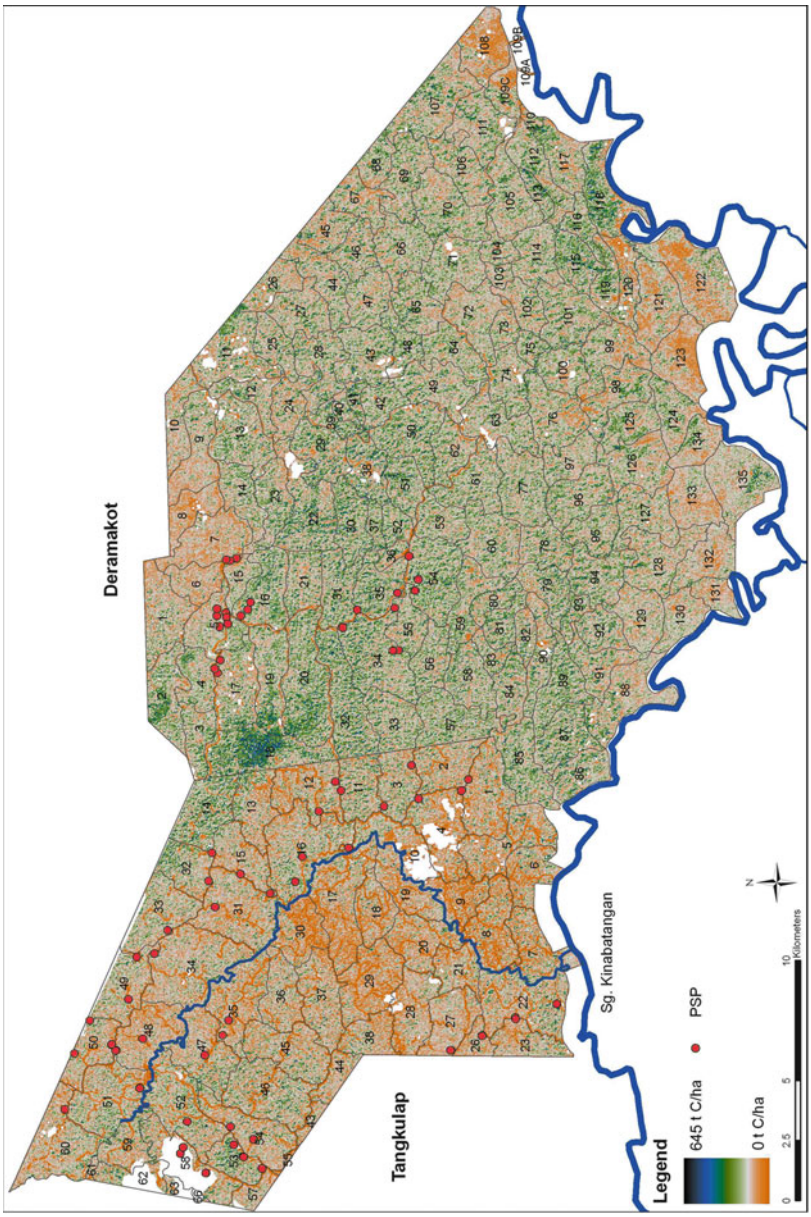


Fig. 2.1 Biomass values per pixel in tons of carbon per hectare (t C/ha) as of the year 2000. *Red dots* symbolize positions of permanent sample plots (*PSP*) and normal labels represent the compartment numbers

A compartment-based analysis of biomass is depicted in Fig. 2.2, showing a great variation in terms of mean biomass among the compartments. The values range from 56 t C/ha in compartment 55 of Tangkulap to 239 t C/ha in compartment 18 of Deramakot. The compartments in the central and western part of Deramakot have the highest above-ground biomass values as of 2000. Lowest biomass values occur in the central compartments of Tangkulap.

However, haze also affected these results. The 2000 Landsat scene showed low levels of haze for almost all compartments of Tangkulap, but only for the northern and southern compartments of Deramakot (Fig. 2.2). Haze leads to an underestimation of biomass, and consequently biomass may be slightly underestimated in both forest reserves. The magnitude of the underestimate appears to be greater in Tangkulap.

As already mentioned, haze shadows also affect the accuracy of the analysis, resulting in an overestimation of the above-ground biomass because shadows reduce the reflectance values of the MidIR band. Some compartments (compartment 14 and 32 of Tangkulap and compartment 18 of Deramakot) appear to have an elevated biomass estimate as a result of haze shadows.

2.3.2 Effects of Different Logging Techniques on Above-Ground Biomass

To derive detailed information about the effects of the different forest management techniques on above-ground biomass, it was necessary to compare the changes in biomass over time. For that purpose, biomass values were estimated for the year 1991 and 2007, using the same model derived from the 2000 Landsat scene.

2.3.2.1 Conventional Logging in Tangkulap (1991–2000)

The time period between 1991 and 2000 corresponds to the practice of conventional logging in Tangkulap. We analyzed the change in above-ground biomass between 1991 and 2000 to estimate the ecological as well as harvesting impacts of conventional logging. Our analysis revealed a net loss of 6.0 t C/ha (5.2–7.0 t C/ha) per year.

The analysis on a compartment basis showed that all compartments of Tangkulap experienced a net loss in biomass between 1991 and 2000, except for two compartments in the northeast (Fig. 2.3). In contrast, the central and southern compartments showed a high loss of above-ground biomass.

However, haze also influenced these estimates. The horizontally striped compartments in Fig. 2.3 experienced a higher haze impact in 1991 than in 2000 (mainly in the eastern compartments of Tangkulap), which led to a minor underestimation of biomass in 1991 and consequently to an underestimation of carbon loss between 1991 and 2000. The impact of haze shadows in compartment

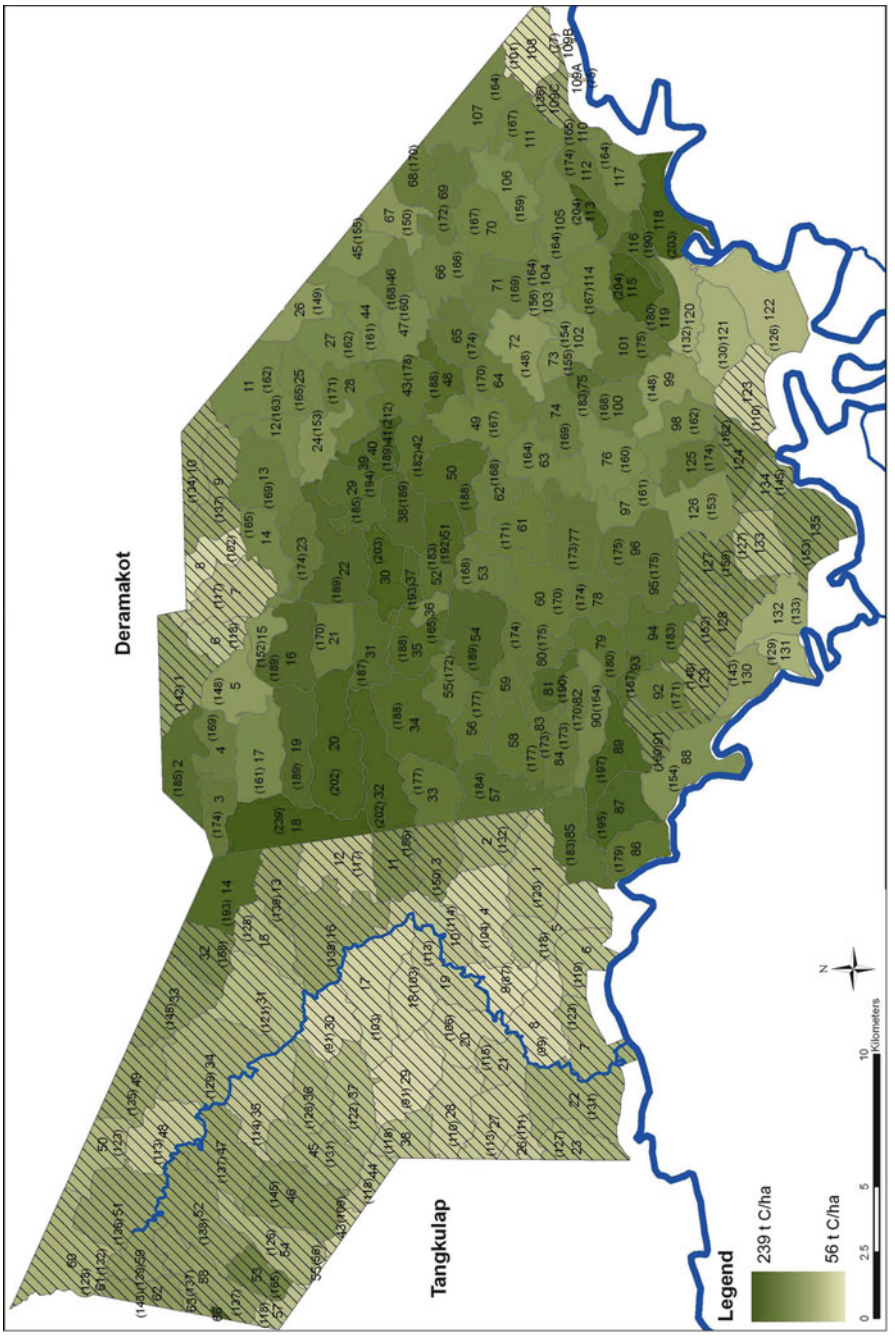


Fig. 2.2 Mean biomass values per compartment (t C/ha) as of the year 2000. Larger font number labels represent the component numbers. Labels in *parentheses* show the average amount of biomass (t C/ha) per compartment. Compartments with haze impacts are displayed with *diagonal stripes*

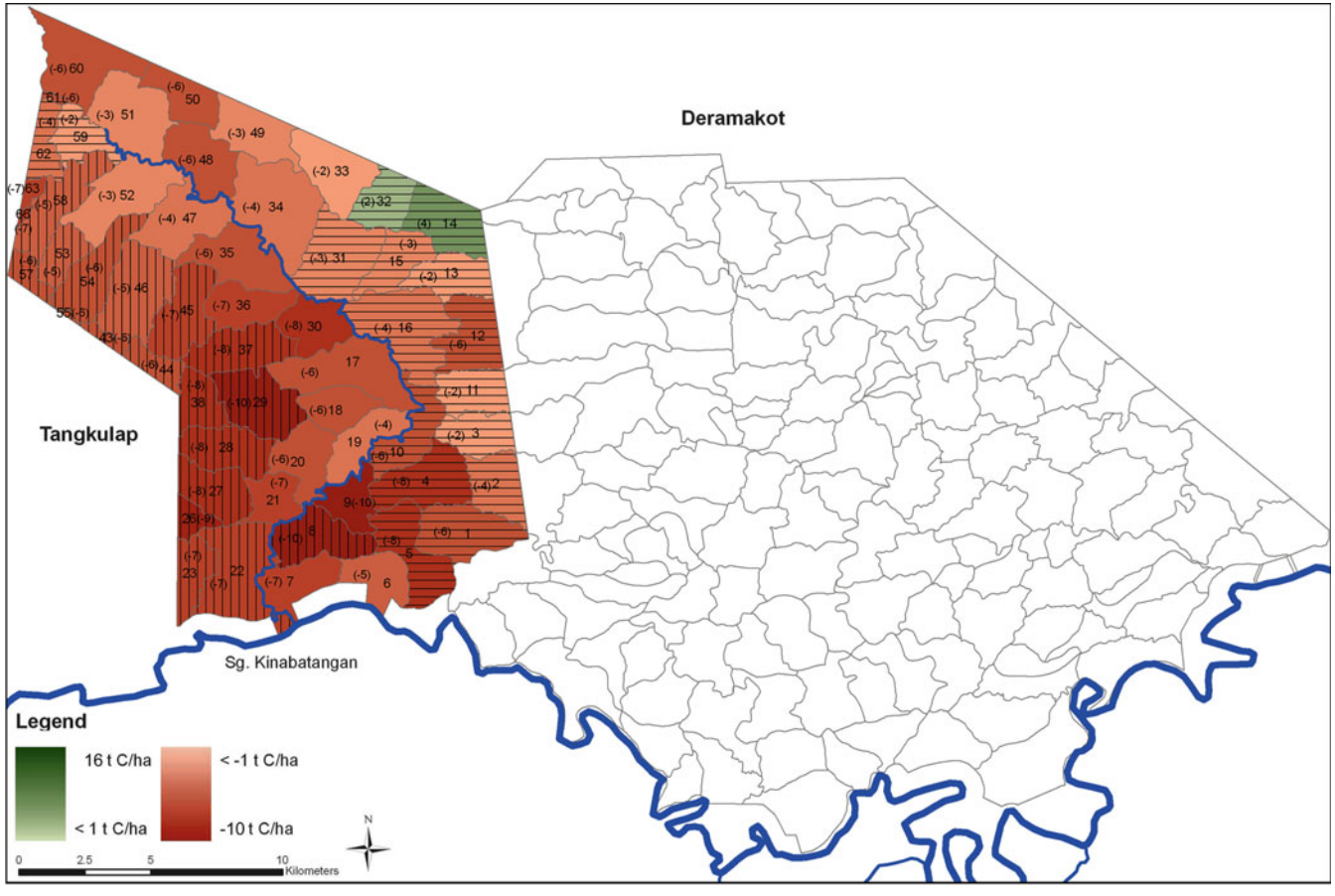


Fig. 2.3 Annual biomass changes (t C/ha) in Tangkulap between 1991 and 2000 (numbers in *parentheses*). Net biomass increase is indicated with *green*; net biomass reduction is indicated with *red*. Compartments with a higher impact of haze in 1991 are displayed with *horizontal stripes*; compartments with *vertical stripes* show a stronger haze impact in 2000. Larger font normal numbers represent compartment numbers

14 and 32 in the year 2000, as referred to previously, additionally strengthened this misinterpretation. Vertically striped compartments (as shown in Fig. 2.3) had a higher impact of haze in 2000, which caused a small underestimation of biomass in that year (in the western compartments), resulting in an overestimation of carbon loss. Compartments with no stripes showed similar impacts of haze in both years (see Table 2.1 for a complete overview of all compartments in Tangkulap).

2.3.2.2 Reduced-Impact Logging in Deramakot (2000–2007)

Almost all compartments of Deramakot demonstrated an increase in above-ground biomass values over the investigation period from 2000 to 2007, during which reduced-impact logging was undertaken in that forest reserve. The increase was particularly prominent in the central and northern compartments of the forest reserve. However, the biomass recruitment in the northern compartments might be overestimated because of the influence of haze in that area, which was stronger in 2000 than in 2007 (Fig. 2.4; see Table 2.2 for a complete overview of all compartments in Deramakot).

The majority of the southern compartments experienced a much smaller increase of above-ground biomass between 2000 and 2007 and some compartments showed no changes or even minor biomass losses. The forests in the southern parts of Deramakot are generally more degraded than in the central compartments and therefore might have experienced lower biomass recruitment because of the abundant climbers and vines that interfere with the regeneration of the forest. Additionally, the influence of haze in that area was much stronger in 2007 than in 2000, which led to an underestimation of biomass recruitment. Thus, these degraded compartments could have actually gained more biomass than depicted in Fig. 2.4.

Even though Deramakot experienced reduced-impact logging activities between 2000 and 2007 (Gobilik et al. 2010), the mean biomass over the whole of Deramakot increased by 6.1 tC/ha (5.3–7.2 tC/ha) per year. This fact emphasizes that the amount of extracted timber and collateral damages by reduced-impact logging did not exceed the capacity of forest regrowth.

2.3.2.3 Comparison of the Two Forest Reserves

Because there is still great variation among the compartments in each forest reserve in terms of haze impact, a compartment-based analysis would show higher inaccuracies than an analysis on a forest management unit level (i.e., forest reserve). The reason is that an analysis on a management unit level integrates over various compartments, differently affected by haze, thus decreasing the overall error. Therefore, when using the afore-described biomass monitoring technique, we strongly

Table 2.1 Above-ground biomass change between 1991 and 2000 of each single compartment in the Tangkulap Forest Reserve

Co	%	Hz	Δ	Co	%	Hz	Δ	Co	%	Hz	Δ	Co	%	Hz	Δ
1	52	−1	−6	**16**	69	−1	−4	**33**	72	0	−2	**52**	69	0	−3
2	34	−2	−4	**17**	73	0	−6	**34**	91	0	−4	**53**	90	1	−5
3	61	−2	−2	**18**	68	0	−6	**35**	84	0	−6	**54**	92	1	−6
4	36	−1	−8	**19**	70	0	−4	**36**	89	0	−7	**55**	79	3	−6
5	79	−1	−8	**20**	67	0	−6	**37**	74	1	−8	**57**	85	2	−6
6	77	0	−5	**21**	78	0	−7	**38**	82	1	−8	**58**	46	1	−5
7	89	0	−7	**22**	91	1	−7	**43**	79	2	−6	**59**	84	−1	−2
8	87	1	−10	**23**	93	3	−7	**44**	73	2	−6	**60**	93	0	−6
9	75	0	−10	**26**	88	4	−9	**45**	87	1	−7	**61**	97	−1	−6
10	44	−1	−6	**27**	78	3	−8	**46**	88	1	−5	**62**	48	−1	−4
11	57	−2	−2	**28**	70	2	−8	**47**	75	0	−4	**63**	38	0	−7
12	66	−1	−6	**29**	84	1	−10	**48**	87	0	−6	**66**	18	2	−7
13	57	−2	−2	**30**	58	0	−8	**49**	82	0	−3				
14	42	−2	4	**31**	72	−1	−3	**50**	87	0	−6				
15	43	−2	−3	**32**	46	−2	2	**51**	89	0	−3				

"Co" represents the compartment number (in bold); "%" shows the percentage of analyzed area per compartment; "Hz" displays the impact of haze ranging from −4 (haze much stronger in 1991 than 2000) over "0" (haze similar in both years) to +4 (haze much weaker in 1991 than 2000); "Δ" indicates the change in above-ground biomass in tons C/ha per year

recommend the analysis and interpretation on a forest management unit level only. In the following, we compared consequences of different managements on a management unit basis.

Tangkulap lost 6.0 t C/ha (5.2–7.0 t C/ha) per year during 1991 and 2000, the time period when timber was extracted using a destructive conventional logging system. Deramakot gained, under reduced-impact logging, an annual amount of 6.1 t C/ha (5.3–7.2 t C/ha) between 2000 and 2007. This result confirms that sustainable forest management with reduced-impact logging, in contrast to conventional logging, helps to conserve above-ground biomass. Comparing these results in absolute terms, we conclude that an amount of at least 10.5 t C/ha per year can be attributed to the different management systems.

Because climate change mitigation is one of the important ecosystem services provided by tropical rain forests for all human beings, emission reduction through sustainable forest management should be rewarded in monetary terms. Such rewards will become a financial incentive to better adopt a sustainable forest management system in production forests. International frameworks and institutions, such as Reducing Emissions from Deforestation and Forest Degradation (REDD+), need to be developed to encourage sustainable forestry, and therefore a transparent monitoring and verification system is crucially needed. Our analysis using

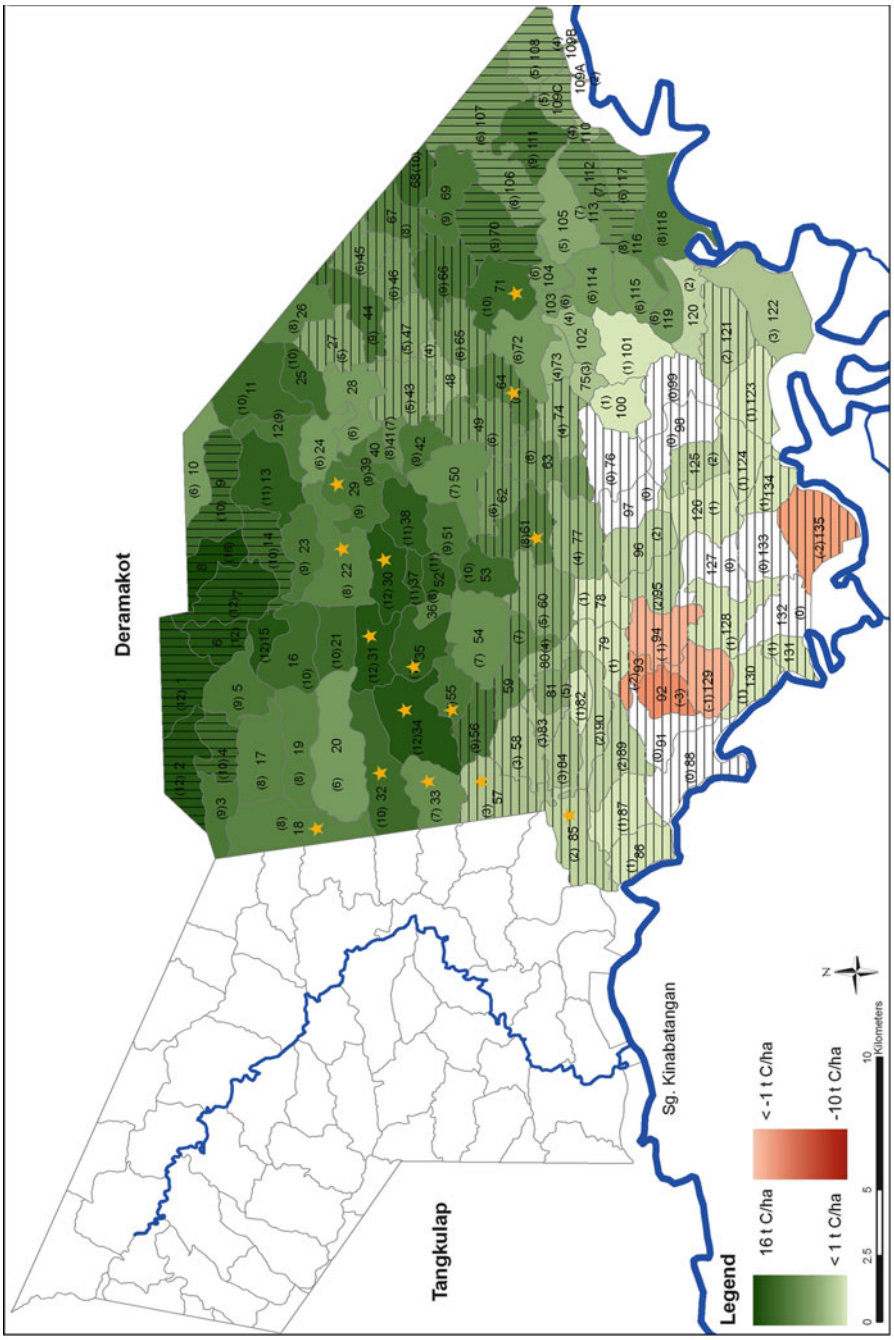


Fig. 2.4 Annual biomass changes (t C/ha) between 2000 and 2007 for Deramakot. Net biomass increase is indicated with *green*; net biomass reduction is indicated with *red*. Compartments with haze are indicated as in Fig. 2.3 except that *horizontal stripes* represent a stronger impact in 2007. Compartments in which reduced-impact logging has been undertaken are marked with *yellow stars*. *Larger font normal numbers* represent compartment numbers

Table 2.2 Above-ground biomass change between 2000 and 2007 of each single compartment in the Deramakot Forest Reserve

Co	%	Hz	Δ	Co	%	Hz	Δ	Co	%	Hz	Δ	Co	%	Hz	Δ
1	96	−2	12	**36**	80	0	8	**71**	82	0	10	**106**	79	−1	6
2	93	−1	12	**37**	88	0	11	**72**	86	0	6	**107**	69	−1	6
3	85	0	9	**38**	82	0	11	**73**	86	0	4	**108**	74	−1	5
4	98	−1	10	**39**	87	0	9	**74**	84	1	4	**109**	77	−1	5
5	95	0	9	**40**	81	0	8	**75**	76	0	3	**109A**	65	−1	2
6	98	−1	12	**41**	87	0	7	**76**	69	1	0	**109B**	65	−2	4
7	93	−2	12	**42**	87	0	9	**77**	74	1	4	**110**	61	−1	4
8	87	−4	16	**43**	82	1	5	**78**	66	3	1	**111**	70	−1	9
9	85	−2	10	**44**	80	1	9	**79**	76	2	1	**112**	79	−1	7
10	75	0	6	**45**	48	1	6	**80**	86	2	4	**113**	39	0	7
11	76	0	10	**46**	71	1	6	**81**	94	2	5	**114**	74	0	6
12	88	0	9	**47**	78	2	5	**82**	83	2	1	**115**	71	0	6
13	83	0	11	**48**	75	2	4	**83**	81	2	3	**116**	84	0	8
14	91	−1	10	**49**	86	1	6	**84**	91	2	3	**117**	84	−1	6
15	98	0	12	**50**	81	0	7	**85**	95	2	2	**118**	75	0	8
16	87	0	10	**51**	84	0	9	**86**	95	1	1	**119**	86	0	6
17	83	0	8	**52**	89	0	11	**87**	93	2	1	**120**	81	0	2
18	68	0	8	**53**	80	0	10	**88**	83	3	0	**121**	78	1	2
19	94	0	8	**54**	91	0	7	**89**	92	1	2	**122**	85	0	3
20	95	0	6	**55**	91	0	9	**90**	84	1	2	**123**	87	1	1
21	83	0	10	**56**	58	1	9	**91**	68	3	0	**124**	86	2	1
22	82	0	8	**57**	73	1	3	**92**	54	3	−3	**125**	65	2	2
23	88	0	9	**58**	34	1	3	**93**	43	3	−2	**126**	81	2	1
24	87	0	6	**59**	81	1	7	**94**	67	3	−1	**127**	79	1	0
25	88	0	10	**60**	86	1	5	**95**	83	2	2	**128**	85	1	1
26	75	0	8	**61**	82	1	8	**96**	70	2	2	**129**	84	2	−1
27	69	1	5	**62**	75	1	6	**97**	66	1	0	**130**	88	2	1
28	78	0	6	**63**	82	1	6	**98**	75	2	0	**131**	91	2	1
29	86	0	9	**64**	88	1	8	**99**	71	1	0	**132**	86	2	0
30	87	0	12	**65**	87	1	6	**100**	69	0	1	**133**	79	1	0
31	93	0	12	**66**	84	1	9	**101**	81	0	1	**134**	81	1	1
32	94	0	10	**67**	83	1	8	**102**	87	0	4	**135**	68	2	−2
33	90	0	7	**68**	83	1	10	**103**	68	0	6				
34	91	0	12	**69**	83	0	9	**104**	81	0	6				
35	89	0	11	**70**	85	−1	9	**105**	65	0	5				

"Co" represents the compartment number (in bold); "%" shows the percentage of analyzed area per compartment; "Hz" displays the impact of haze ranging from -4 (haze much stronger in 2000 than 2007) over "0" (haze similar in both years) to +4 (haze much weaker in 2000 than 2007); "Δ" indicates the change in above-ground biomass in tons C/ha per year

satellite remote sensing could successfully evaluate the positive effects of sustainable forest management in terms of carbon increment on a forest management unit level.

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Chapter 3
Management Effects on Tree Species Diversity and Dipterocarp Regeneration

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3.1 Introduction

Unregulated selective logging has been common throughout the Southeast Asian tropical rain forests. Such logging usually damages more than 50% of the original forest biomass and causes soil disturbances by the use of heavy machinery (Cannon et al. 1994; Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998). To mitigate the deleterious logging impacts, reduced-impact logging has been applied in some tropical production forests (Kleine and Heuveldop 1993; Lagan et al. 2007; Putz et al. 2008a). Reduced-impact logging is a modification of selective logging that is intended to minimize collateral damage to the residual stands. Such methods include pre-harvest inventory, mapping of all canopy trees, directional felling, liana cutting, and planning of skid trails, log decks, and roads (see Chap. 1). In comparison with

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unregulated conventional logging, reduced-impact logging is beneficial in maintaining not only future crop trees (Rockwell et al. 2007; Peña-Claros et al. 2008) and forest biomass (Johns et al. 1996; Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998; Putz et al. 2008b; Imai et al. 2009), but also biological diversity, such as dung beetles (Davis 2000), flying insects (Akutsu et al. 2007), soil fauna (see Chap. 4 by Hasegawa et al., this volume), and forest-dwelling vertebrates (see Chap. 5 by Samejima et al., this volume). Many other taxa, such as ants, arachnids, bats, birds, fishes, and animals, also are not adversely affected by reduced-impact logging (Azevedo-Ramos et al. 2006; Wunderle et al. 2006; Castro-Arellano et al. 2007; Felton et al. 2008; Presley et al. 2008; Dias et al. 2010; Bicknell and Peres 2010). However, the effects of reduced-impact logging on tree species diversity of tropical rain forests remain largely unknown, despite that the diversity of trees is fundamental to the structure and functions of the forests. The diversity of trees may also determine the diversity of other taxonomic groups because trees provide resources and habitat structures for dependent species. The aim of this study is to examine the effects of reduced-impact logging versus conventional logging on tree species diversity and composition in the lowland tropical rain forests in Deramakot and Tangkulap.

Previous work has showed inconsistent responses of tree species diversity to selective logging, with unchanged (Slik et al. 2002; Verburg and van Eijk-Bos 2003; Bischoff et al. 2005), decreased (Okuda et al. 2003; Brearley et al. 2004; Makana and Thomas 2006), and increased (Plumptre 1996; Cannon et al. 1998; Berry et al. 2010) diversity following logging. One possible factor for such inconsistent responses is the variation of logging intensity at which tree diversity was measured among different studies. In the course of secondary succession, dominant species may change from early-successional pioneers to late-successional climax species. The intermediate disturbance hypothesis (Connell 1978), which postulates a unimodal relationship of species diversity along a gradient of disturbance, predicts that tree diversity declines following a severe habitat disturbance, whereas tree diversity may increase following a moderate habitat disturbance resulting from the coexistence of pioneer and climax species. This theory implies that tree diversity may decline following heavy selective logging, although it may increase following selective logging with moderate intensity. Selective logging actually has a wide variation in the volume of timber extracted and the severity of damage caused by the removal of individual trees and road construction (Cannon et al. 1994). However, previous research has compared tree diversity between unlogged and logged forests without considering variation of logging intensity (Cannon et al. 1998; Okuda et al. 2003; Brearley et al. 2004; Bischoff et al. 2005; Makana and Thomas 2006; Barlow et al. 2007; Berry et al. 2008; but see Verburg and van Eijk-Bos 2003). Such uncontrolled logging intensity may thus explain the inconsistent results among the previous studies. To overcome such a problem, we compared tree assemblage among a primary forest and two logged forests with a contrasting logging intensity, that is, reduced-impact logged (moderately logged) forest and conventionally logged (heavily logged) forest.

Diversity following disturbances also depends on the spatial scale at which the diversity are measured (Hill and Hamer 2004; Dumbrell et al. 2008). Logged-over tropical rain forests consist of a mosaic of patches affected by different intensities

of disturbance, such as unaffected areas, gaps from harvesting and road construction, and regenerating patches. At a large spatial scale, all these patches can be encountered, and thus the species diversity in the area (i.e., α-diversity) may become larger than (or similar to) that of primary forests, because logging creates new habitats for species not found in primary forests. At a small spatial scale, only one or a few patches can be encountered. If sampling was biased toward degraded patches with few recruitments or the patches dominated by a few pioneer species, species diversity becomes lower in logged-over forests than in primary forests. Accordingly, spatial scale, which determines the number of different patch types encountered in a given area, must affect tree diversity in that area. However, previous studies assessing tree diversity in degraded tropical rain forests have used small sample plots (approximately <0.2 ha in area), and only a few studies have taken into account the effects of different spatial scale on tree diversity (but see Berry et al. 2008). We therefore established both small (0.2-ha) and large (2-ha) sample plots in each of the three forests and compared tree diversity at four different scales: species richness within a 0.2-ha plot (i.e., α-diversity at 0.2-ha scale including only a few patches), species richness within a 2-ha plot (i.e., α-diversity at 2-ha scale including multiple patches), species turnover within a 2-ha plot (i.e., β-diversity at 2-ha scale), and species turnover among replicated 0.2-ha plots in each of the three forests (i.e., β-diversity among the stands that may include limited types of patches).

Species accumulation curves (species–area and species–individual curves) and multivariate techniques (e.g., ordination) were used to study species diversity and community similarity, respectively, in our species-rich tropical rain forests. The documentation of how species richness increases with the area censused or with the number of trees censused is crucial for understanding species diversity (Condit et al. 1996; Gotelli and Colwell 2001). In addition, species composition and community similarity may be more sensitive to disturbances or land-use changes than is species richness (Su et al. 2004; Barlow et al. 2007). We examined the impacts of varying logging intensities on tree species diversity and composition by a combination of multivariate technique (i.e., nonmetric multidimensional scaling), species accumulation curves, and comparison of species diversity at different spatial scales.

Bornean lowland tropical rain forests are dominated by Dipterocarpaceae, which family contains many species important as marketable timber. Selective logging including reduced-impact logging usually harvests large (≥60 cm in diameter in Sabah and ≥40 cm in other areas) dipterocarp trees, but inevitably disturbs the smaller trees also. The regeneration condition of dipterocarp species following logging influences the diversity of dipterocarp species and the future timber yield of the forest. Bischoff et al. (2005) studied the effects of conventional logging on dipterocarp recruitment. Pinard et al. (2000) documented plant recovery (including dipterocarp species) only on skid trails following conventional logging and reduced-impact logging. However, the effects of reduced-impact logging on density and recruitment of dipterocarp seedlings at a large scale (e.g., >1 ha) have not yet been evaluated. We therefore evaluated the effects of reduced-impact logging on the regeneration of dipterocarp species by comparing three 2-ha plots.

3.2 Study Site

This study was carried out in the Deramakot Forest Reserve and the Tangkulap Forest Reserve in Sabah, Malaysia (5°14–30'N, 117°11–36'E). The mean annual temperature of the area is 27°C and the mean annual precipitation is about 3,500 mm, with little seasonal variation. Altitude in the reserves ranges between 20 and 300 m above sea level, and the entire area is covered with lowland mixed dipterocarp forests. Deramakot (551 km^2) and Tangkulap (275 km^2) are located adjacent to each other. Deramakot is a sustainably managed forest with reduced-impact logging, whereas Tangkulap is a forest repeatedly logged using a conventional logging technique. According to the estimation of disturbance intensity in the two reserves using Landsat satellite data (see Chap. 1), much heavier disturbances have occurred in greater areas in Tangkulap compared with Deramakot as a result of unregulated conventional logging during the period from the mid-1980s to 2000. Site conditions and management histories of the two reserves are described in more detail in Chap. 1.

Eleven research plots, 0.2 ha in area, were established in Deramakot and Tangkulap during May 2003–March 2005 (Seino et al. 2005). In Deramakot, four plots were established in a primary forest that had not been logged since the 1970s, and four plots were set up in the forest logged by reduced-impact logging during 1995–2000. In Tangkulap, three plots were established in the forest logged by the conventional method. The three conventionally logged plots were once logged before 1991, and again logged more heavily during 1991–1999, based on the observation of Landsat scenes (A. Langner, personal communication). Eleven 0.2-ha plots were laid out primarily as 100 m length × 20 m width, except as 50 × 40 m in three of the four plots in the primary forest and one of the three plots in the conventionally logged forest because of the limited availability of uniform topography. Mean distances among the 0.2-ha plots are 435 ± 209, 1,130 ± 646, and 756 ± 175 m for the primary, the reduced-impact logged, and the conventionally logged forests, respectively. During November 2006–February 2008, we enlarged one representative 0.2-ha reduced-impact logged plot and one representative 0.2-ha conventionally logged plot to 2 ha in area (200 × 100 m). We additionally established a new 2-ha plot in the primary forest, because the four 0.2-ha primary plots are in small patches of a residual forest and therefore cannot be enlarged without including degraded patches. Each of the 2-ha plots consists of 10 × 10 m subplots.

3.3 Tree Species Diversity

First, to describe forest structure and species composition, all trees ≥ 10 cm diameter at breast height (dbh) were measured in each plot. We also established one hundred 5 × 5 m plots within a 2-ha plot (allocated alternately to each 10 × 10 m subplot) and measured the dbh of small trees with 5–10 cm dbh (Fig. 3.1). Buttressed trees were measured at well above (~50 cm) protrusions. Above-ground biomass was estimated according to the allometric equation obtained by Brown (1997) as

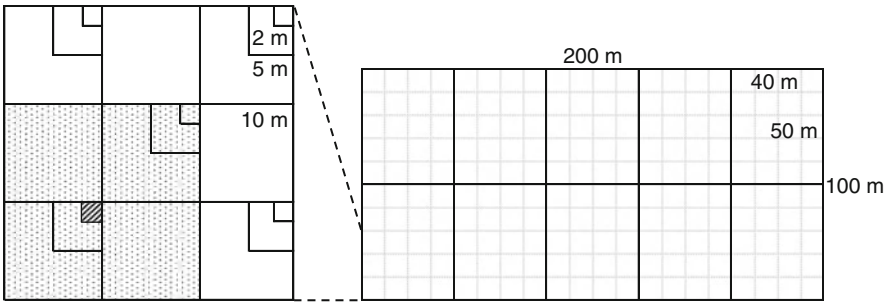


Fig. 3.1 A 2-ha (200 × 100 m) plot was established in each of the three forests. A 2-ha plot consists of 200 10×10 m subplots. One hundred 5×5 m plots and 100 2×2 m plots were further established within a 2-ha plot. Plots smaller than 5×5 m were allocated alternately to each of the 10×10 m subplots. Stems were tagged and measured in plots as follows: trees ≥10 cm dbh in 10×10 m plot, trees ≥5 cm dbh and dipterocarp saplings (1–5 cm dbh) in 5×5 m plot, and dipterocarp seedlings (≥10 cm high and <1 cm dbh) in 2×2 m plot. Species composition at the 40×50 m scale within a 2-ha plot (i.e., ten 0.2-ha subplots per 2-ha plot) was used to study the variation of tree species diversity and composition within 2 ha (*right figure*). Basal area (BA) of dipterocarp species ≥10 cm dbh was analyzed at the 20×20 m scale (*dotted area, left figure*) because it was considered as a factor affecting the density of dipterocarp seedlings in the corresponding 2×2 m plot (*hatched area*), which is located near the center of a 20×20 m subplot

$$\text{Above}-\text{ground biomass}\,(\text{kg ha}^{-1}) = \exp(-2.134 + 2.530\ \ln\text{dbh})$$

Voucher specimens were collected from the trees that could not be identified in situ. Specimens taken were identified at the Herbarium, Forest Research Centre, Sabah Forestry Department, Sandakan, Sabah. Samples that could not be identified to species were distinguished as morpho-species. Species diversity was evaluated by three measures as follows: species richness (S) in terms of the number of species per plot; Fisher's α, $S = \alpha \ln(1+N/\alpha)$, which is commonly used for highly diverse tropical species data (Condit et al. 1996) (where S is the number of species, N is the number of individuals, and ln is natural logarithm); and Shannon's index:

$$H' = -\sum_{i=1}^{s} p_i \log p_i$$

where s is the number of species and p_i is the relative dominance of the ith species based on relative stem number. Species diversity in an entire landscape (γ-diversity) is related to the species composition within each of the segments of the landscape (α-diversity) and the difference in species composition among those segments (β-diversity) (Whittaker 1972). We estimated β-diversity within a 2-ha plot and among 0.2-ha plots in each forest type by the similarity difference in species composition between a given pair of plots in each forest type (Vellend 2001), using the Morisita's similarity index (C_λ) (Morisita 1959):

$$\beta = 1 - C_\lambda$$

Similarity index (C_λ) ranged from 0 (completely different between two communities) to 1 (completely the same). For the 0.2-ha plots, β was calculated between two plots in each forest type (six pairs each in the primary and the reduced-impact logged forests and three pairs in the conventionally logged forest). For the 2-ha plots, we utilized species composition at the scale of 40×50 m within each of the 2-ha plots (i.e., ten 0.2-ha subplots by sequentially shifting a 40×50 m quadrat per 2-ha plot) (Fig. 3.1), and then β was calculated for 45 pairs of subplots in each 2-ha plot.

Species accumulation curves were constructed by using EstimateS (statistical estimation of species richness and shared species from samples, ver. 8; http://viceroy.eeb.uconn.edu/estimates). A species–area curve is the mean of 100 randomizations when 200 10×10 m plots for trees ≥10 cm dbh and 100 5×5 m plots for trees with 5–10 cm dbh in each 2-ha plot were sequentially and randomly added to increase the area. The species–individual curves were obtained by converting a cumulative area to the cumulative number of stems.

Difference in the vegetation properties (above-ground biomass, basal area, stem density, maximum dbh, and species diversity indices) among forest types was tested by an analysis of variance (ANOVA). When the ANOVA P value was <0.05, the Tukey–Kramer post hoc test was performed to determine which pairs of means differed significantly.

Table 3.1 shows the forest structural features in the three forests (mean values of one 2-ha plot and two to four 0.2-ha plots per forest type). Estimated above-ground biomass, total basal area, and maximum dbh were significantly greater in the reduced-impact logged forest than in the conventionally logged forest but did not differ between the reduced-impact logged and the primary forest. Stem density for trees ≥10 cm dbh did not differ among forest types (Table 3.1). However, the density of large trees ≥60 cm dbh (which demarcates a minimum dbh for logging) was significantly lower in the conventionally logged forest than in the other two forests (Fig. 3.2; $P<0.05$ for 0.2-ha plots). The density of small trees with 5–10 cm dbh was also significantly lower in the conventionally logged forest than in the other two forests (Fig. 3.2; $P<0.05$ for 2-ha plot).

In the eleven 0.2-ha plots (in total, 2.2 ha in area), a total of 360 species were recorded among 1,324 stems ≥10 cm dbh. Number of families and genera, and three measures of species diversity (species richness, H', and α), were significantly greater in the reduced-impact logged forest than in the conventionally logged forest but did not differ between the reduced-impact logged and the primary forest (Table 3.2). Among the eleven 0.2-ha plots, species richness, H', and α were positively correlated with above-ground biomass ($r^2=0.69$, 0.53, and 0.72, respectively; all $P<0.05$). In the three 2-ha plots (in total, 6 ha in area), a total of 589 species were recorded among 3,614 stems ≥5 cm dbh (544 species among 2,992 stems ≥10 cm dbh). In the 2-ha plots, the number of families and genera seemed to be similar among forest types. Three measures of species diversity appeared to be higher in the reduced-impact logged forest than in the primary forest. Three measures of species diversity in the conventionally logged forest seemed to be lower than those in the other two forests, as the 0.2-ha plots demonstrated. However, the magnitude of the

Table 3.1 Forest structural features of the three forests

	Primary		Reduced-impact		Conventional	
Maximum dbh (cm)	116.7[a]	± 12.1	119.0[a]	± 17.2	69.6[b]	± 19.8
Basal area (m^2 ha^{-1})						
Total	40.2[a]	± 5.1	33.1[a]	± 4.9	18.8[b]	± 6.5
Dipterocarp species	19.5[a]	± 3.4	13.0[b]	± 3.4	5.9[c]	± 2.3
Pioneer species	1.1[b]	± 0.6	2.4[a,b]	± 2.0	6.0[a]	± 3.3
Above-ground biomass (t ha^{-1})	484.0[a]	± 71.4	381.7[a]	± 63.8	182.2[b]	± 78.1
Stem density (ha^{-1})	649.9	± 92	625.4	± 107	494.0	± 114

Maximum dbh, basal area, above-ground biomass, and stem density for trees ≥ 10 cm dbh are shown. Mean (±SD) values of one 2-ha plot and two to four 0.2-ha plots (four, three, and two plots in the primary, the reduced-impact logged, and the conventionally logged forest, respectively) per forest type are shown. Forests sharing the same letters do not differ significantly at $P<0.05$. Measures without superscript (e.g., above-ground biomass) shows no significant difference ($P>0.05$ between forest types by ANOVA.

Pioneer species: five *Macaranga* species (*M. conifera*, *M. gigantea*, *M. hypoleuca*, *M. pearsonii*, *M. bancana*) and two *Croton* species (*C. argyratus*, *C. oblongus*) of Euphorbiaceae and two Rubiaceae species (*Neolamarckia cadamba*, *Neonauclea* sp.)

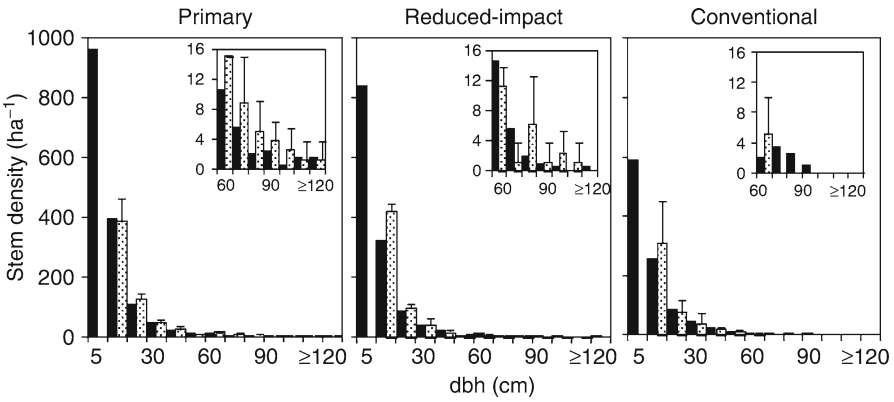


Fig. 3.2 Frequency distribution of diameter at breast height (dbh) for trees ≥5 cm (*larger graphs*) and ≥60 cm dbh (*smaller graphs*) in the three forests. *Black* and *dotted bars* indicate values of 2 and 0.2-ha plots, respectively. *Error bars* indicate SD

differences of diversity between the conventionally logged forest and the other two forests were smaller in the 2-ha plots than in the 0.2-ha plots. For example, species richness of the primary and the reduced-impact logged forest was twofold greater than that of the conventionally logged forest in the 0.2-ha plots, although only 1.2 times greater in the 2-ha plots.

Species–area curves graphically suggested that species richness at a comparable area was greater in the reduced-impact logged forest than in the conventionally logged forest, but richness did not differ greatly between the reduced-impact logged and the primary forest, regardless of tree size class and plot size (Fig. 3.3a–c, g–j). However, species–individual curves for small trees (5–10 cm and 10–20 cm dbh)

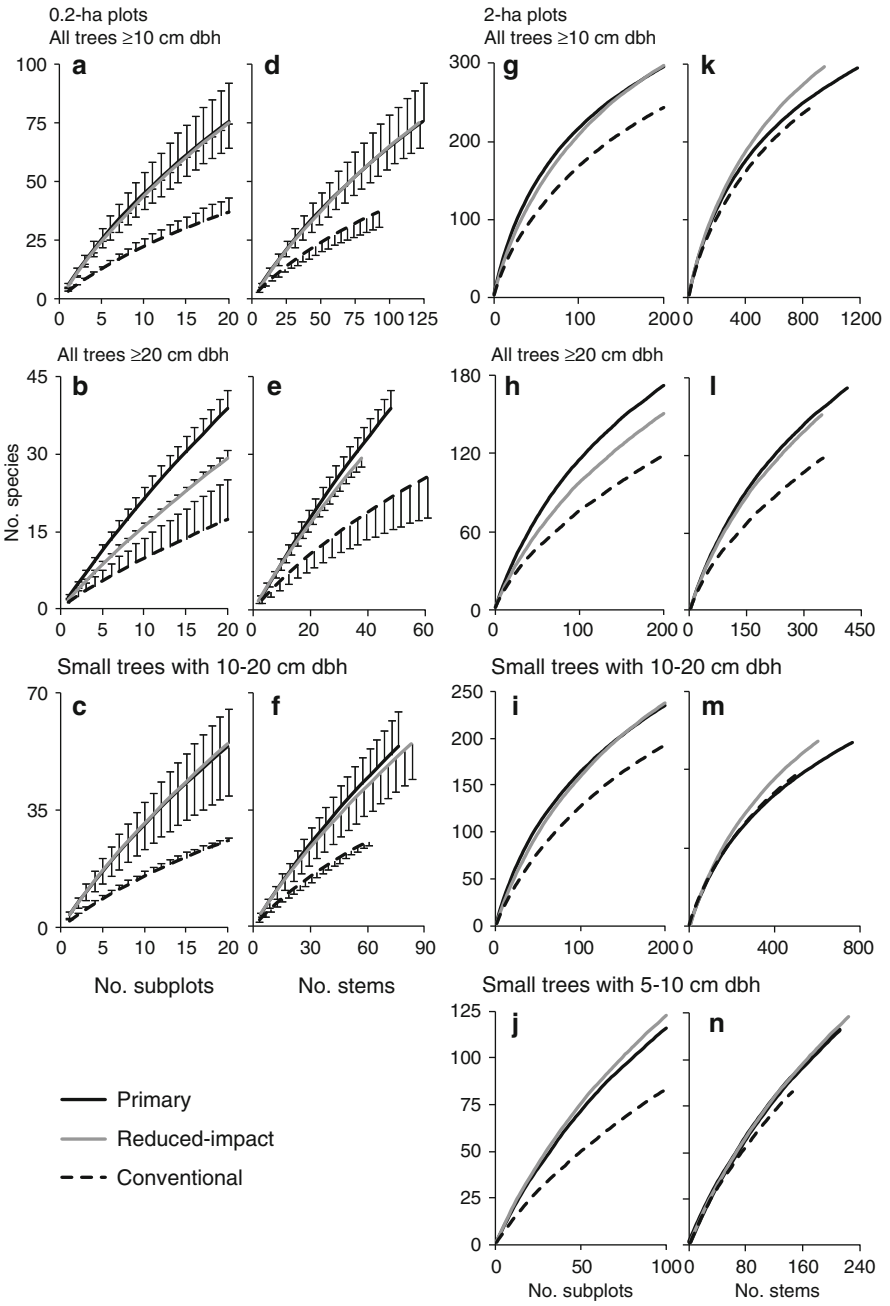


Fig. 3.3 Species accumulation curves by tree size class in 0.2-ha plots (**a–f**) and 2-ha plots (**g–n**). *Error bars* indicate SD. Species–area curves, **a–c** and **g–j**; species–individual curves, **d–f** and **k–n**

Table 3.2 Species diversity for trees ≥ 10 cm dbh (for trees ≥ 5 cm dbh in parentheses) in the three forests

	No. of families		No. of genera		Species richness		Shannon's index *H'*		Fisher's α	
0.2-ha plots										
Primary	30.3[a]	± 1.9	51.5[a]	± 5.4	82.5[a]	± 9.3	6.07[a]	± 0.21	98.3[a]	± 20.0
Reduced-impact	29.0[a]	± 2.0	51.5[a]	± 4.5	79.5[a]	± 10.3	5.83[a]	± 0.35	88.9[a]	± 24.1
Conventional	20.7[b]	± 2.1	30.0[b]	± 4.6	39.3[b]	± 6.4	4.16[b]	± 0.55	26.0[b]	± 9.1
2-ha plots										
Primary	52	(52)	135	(144)	296	(319)	7.26	(7.31)	127	(129)
Reduced-impact	51	(53)	124	(134)	298	(340)	7.42	(7.55)	149	(157)
Conventional	48	(51)	121	(131)	243	(263)	6.89	(6.98)	114	(116)

Number of families and genera and species diversity indices are shown
Forests sharing the same letters do not differ significantly at $P < 0.05$. Measures without superscript (e.g., above-ground biomass) shows no significant difference ($P > 0.05$ between forest types by ANOVA.

and trees including small trees (≥10 cm dbh) in 2-ha plots (Fig. 3.3k,m,n) did not differ from each other among forest types. This finding indicates that the lower species richness of small trees in the 2-ha conventionally logged plot was the result of the lower number of stems per area but not the lower number of species per unit stem number. In fact, the density of small trees (5–20 cm dbh) in the 2-ha conventionally logged plot was lower than those in the other two plots (1,360, 1,162, and 843 trees ha^{-1} in the primary, the reduced-impact logged, and the conventionally logged forest, respectively), but the number of species per unit stem number was similar between 2-ha plots (0.53, 0.50, and 0.56 for trees 5–10 cm dbh and 0.30, 0.37, and 0.38 for trees 10–20 cm dbh in the primary, the reduced-impact logged, and the conventionally logged forest, respectively).

Accordingly, the relationship between α-diversity and logging intensity varied with the spatial scale of sampling. The α-diversity of the primary and the reduced-impact logged forest was twofold greater than that of the conventionally logged forest at a small (0.2-ha) scale but only 1.2 times greater at a medium (2-ha) scale. There was no difference in α-diversity between the reduced-impact logged and the primary forests at a small (0.2-ha) scale, but α-diversity was rather higher in the reduced-impact logged forest than in the primary forest at a medium (2-ha) scale. Logged forests consist of both patches with low diversity (caused by the degradation or the dominance of a few pioneers recruited) and patches with relatively high diversity (caused by the coexistence of small-sized pioneer and climax species). In degraded forests, the latter patches may occupy a small area and distribute heterogeneously. Such patches with high diversity could be encountered at a medium (2-ha) scale, although they could not be adequately sampled at a small (0.2-ha) scale; this may be the reason why the logged forests showed relatively larger α-diversity at the medium (2-ha) scale compared with that at the small (0.2-ha) scale (Table 3.2).

Reflecting such a patchy distribution of tree species in logged-over forests, β-diversity within a 2-ha plot increased with increasing logging intensity (Fig. 3.4a). On the other hand, β-diversity among replicated 0.2-ha plots in each forest type was lower in the conventionally logged forest than in the other two forests; *P* values for

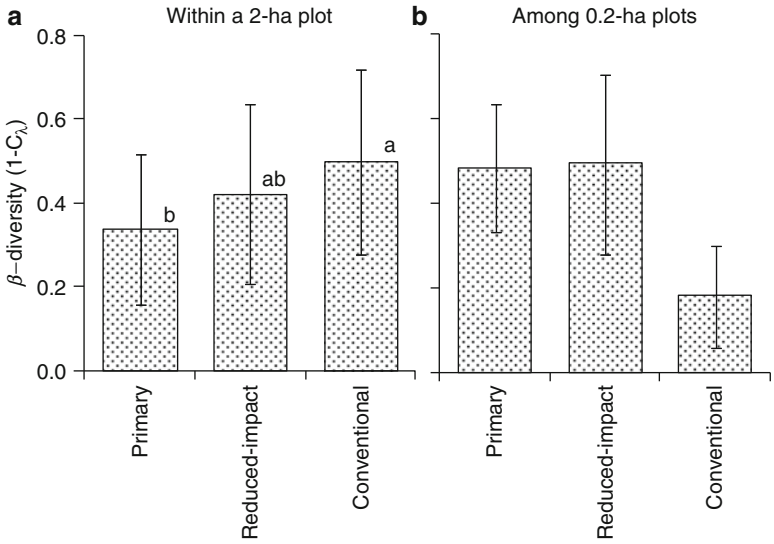


Fig. 3.4 Mean (±SD as error bars) β-diversity, as measured with the dissimilarity of species composition ($1 - C_\lambda$), within a 2-ha plot (**a**) and among replicated 0.2-ha plots (**b**) per forest type. Forests sharing the same letters do not differ significantly at $P < 0.05$. C_λ, Morisita's similarity index

the difference between the conventionally logged and the primary forest, and between the conventionally logged and the reduced-impact logged forests were 0.074 and 0.066, respectively (Fig. 3.4b). Recruitment of pioneer species following logging may increase α-diversity at a medium scale, as already mentioned. However, in general, a few specific species of pioneers (e.g., *Macaranga* spp.) dominate monotonously over a logged landscape. Similarity of species composition within logged-over forests therefore may become high, corresponding to the decreased β-diversity among the 0.2-ha plots in the conventionally logged forest. Accordingly, species diversity is highly dependent upon the spatial scale.

Berry et al. (2008) found no difference in α-diversity between logged and unlogged forests, with higher β-diversity in logged than in unlogged forests, and concluded that selective logging can retain considerable conservation value in remnant forests. Their study was conducted nearby in a forest reserve of Danum Valley, Sabah, with the same vegetation type as ours. Our similar α-diversity between the reduced-impact logged and the primary forests at small (0.2-ha) scale and increased β-diversity (within a 2-ha plot) in the logged forests is consistent with the results of Berry et al. (2008). However, we also found that the conventionally logged forest showed the lowest α-diversity and the lowest β-diversity (among replicated 0.2-ha plots) among forest types. These observations indicate that species diversity is also highly dependent on logging intensity and that high-impact conventional logging indeed can decrease the α-diversity of canopy tree species. On the other hand, reduced-impact logging can be an adequate method to maintain tree diversity at a similar level with primary forest.

3.4 Tree Species Composition

To study the effects of reduced-impact logging on tree species composition, we used a nonmetric multidimensional scaling (NMDS) (a multivariate technique) and examined the similarity of species composition among the three forests. For comparison of species composition among the eleven 0.2-ha plots, NMDS was applied to the relative basal area of species of each plot. For comparison among the three 2-ha plots, the relative basal area of species within 40 × 50-m subplots in each 2-ha plot was used (i.e., ten 0.2-ha subplots in each 2-ha plot) (Fig. 3.1). Sørensen (Bray–Curtis) distance was used to calculate the dissimilarity (distance) of species composition between plots, and a distance matrix was developed. Ordinations were performed on the distance matrix by NMDS for the first six dimensions with 100 random starting configurations using the statistical software package PC-ORD (ver. 4.0; MJM Software). The variation in species composition among the plots was illustrated graphically. We also compared the similarity of species composition between two forests among the three forest types (i.e., primary vs. reduced-impact logged forest, primary vs. conventionally logged forest, and reduced-impact logged vs. conventionally logged forest) by tree size class, using Morisita's index.

NMDS ordinations indicated that species composition of trees ≥10 cm dbh obviously differed between the conventionally logged forest and the other two forests irrespective of plot size but did not differ between the primary and the reduced-impact logged forest (Fig. 3.5a,b). This pattern also persisted in large trees ≥20 cm dbh in the 0.2-ha plots (Fig. 3.5c). For large trees ≥20 cm dbh in the 2-ha plots, the primary and the conventionally logged forests were plotted at the opposite extremes along axis 1, and the reduced-impact logged forest was plotted at an intermediate position (Fig. 3.5d). Species composition of small trees of 5–10 cm dbh in 2-ha plots differed among forest types without much overlap (Fig. 3.5e).

As Fig. 3.5a–d demonstrates, species composition clearly differed between the primary forest and the conventionally logged forest irrespective of tree size (C_λ range, 0.13–0.27) (Fig. 3.6a). Species composition of the reduced-impact logged and the conventionally logged forests also differed in the medium-sized trees but was similar in the small- and large-sized trees (C_λ = 0.45 and 0.79 for trees with 5–10 cm and ≥30 cm dbh, respectively) (Fig. 3.6c). Similarity of species composition in primary and reduced-impact logged forest decreased in small- and large-sized trees (Fig. 3.6b).

To examine the continuous regeneration in terms of community composition within a forest, we calculated the similarity of species composition between small- and large-sized trees (i.e., 5–10 cm vs. ≥10 cm dbh, 5–10 cm vs. ≥20 cm dbh, and 10–20 cm vs. ≥20 cm dbh) within a forest. In the primary forest, species composition was similar between small- and large-sized trees within a forest (C_λ range, 0.75–0.99) (Fig. 3.6d–f). However, in the conventionally logged forest, species composition differed between small- and large-sized trees (C_λ range, 0.24–0.59). The reduced-impact logged forest showed an intermediate pattern (C_λ range, 0.49–0.85). These data indicated that species composition of the primary forest differed considerably from that of the conventionally logged forest, and the reduced-impact logged forest had generally similar species composition with the primary forest but contained

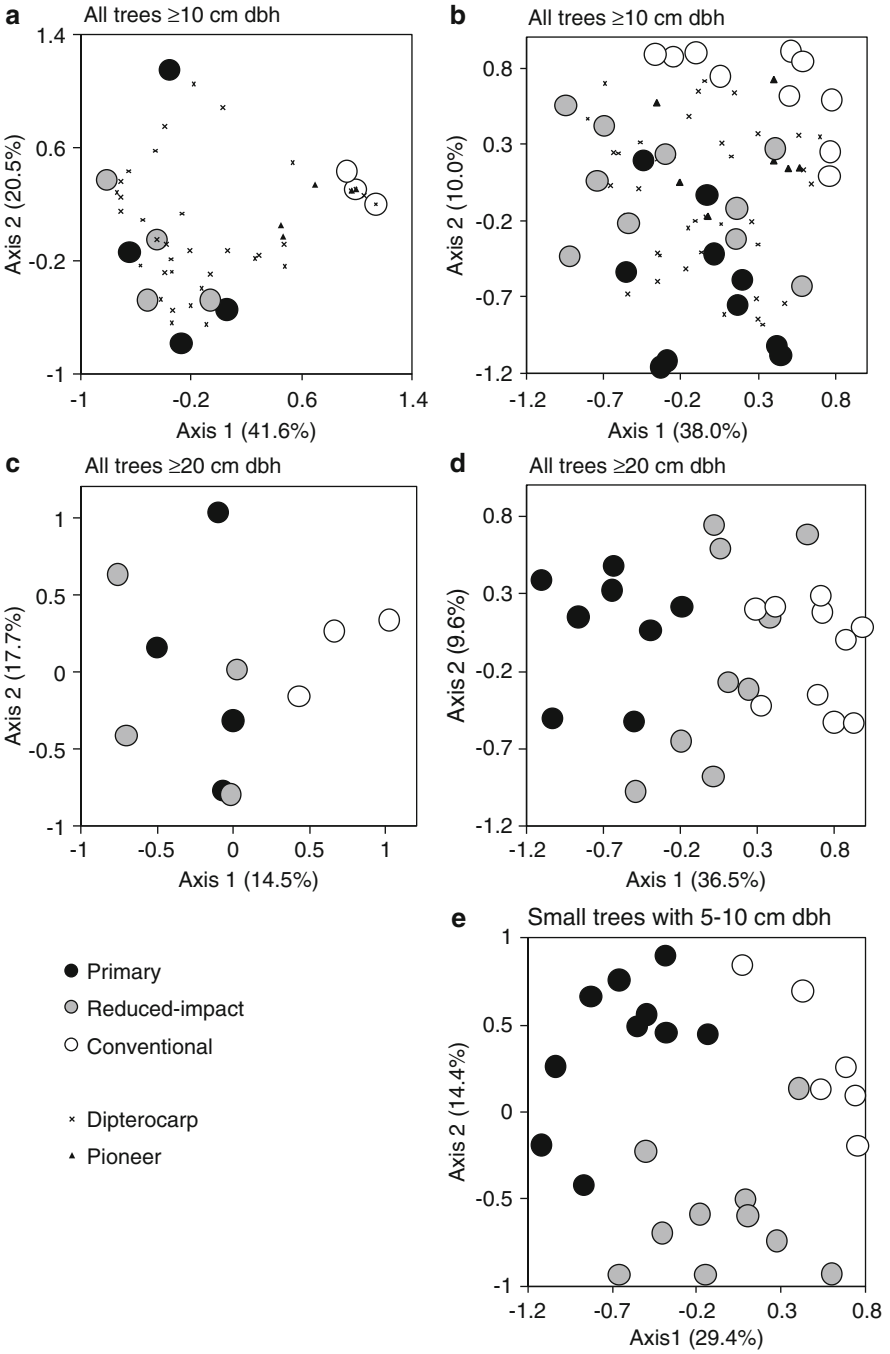


Fig. 3.5 An ordination of the 11 0.2-ha plots (**a**, **c**) and 30 0.2-ha subplots in the three 2-ha plots (**b**, **d**, **e**) by tree size classes (**a**, **b**: ≥10 cm dbh; **c**, **d**: ≥20 cm dbh; **e**: 5–10 cm dbh) on the coordinate of axis 1 and axis 2 of nonmetric multidimensional scaling (NMDS) analysis. Species scores (dipterocarp and pioneer species) are also plotted in diagrams **a** and **b**. Percentage in *parentheses* on each axis is the variation of species composition among plot, which is explained by the analysis

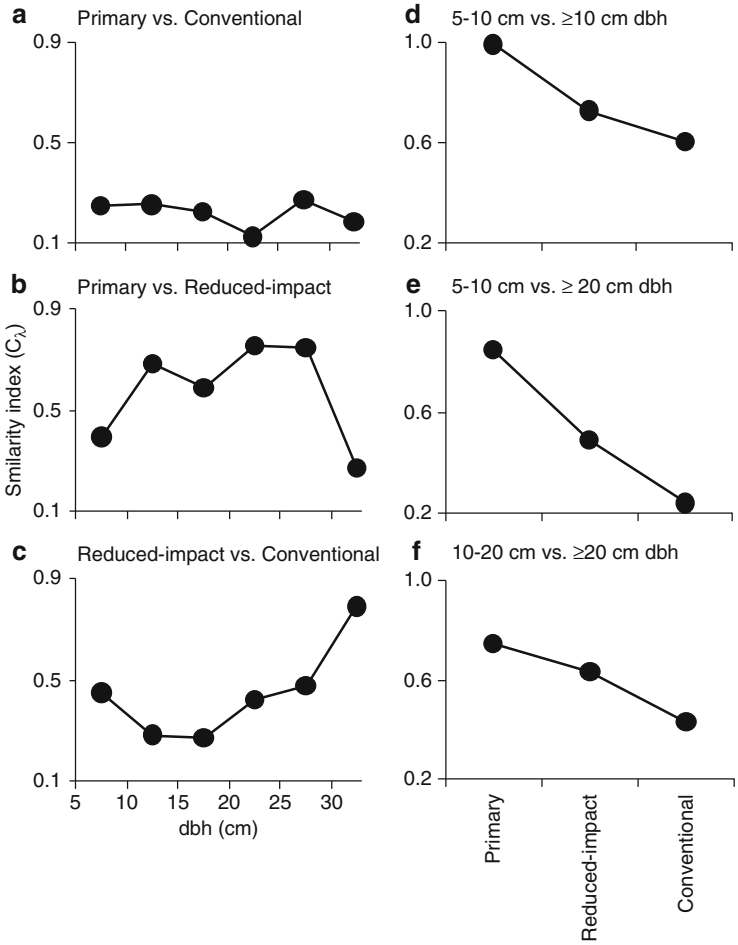


Fig. 3.6 Similarity of species composition (Morisita's C_λ) between forest types at a given size class based on three 2-ha plots (**a–c**) and between pairs of smaller and larger trees within a given forest type (**d–f**)

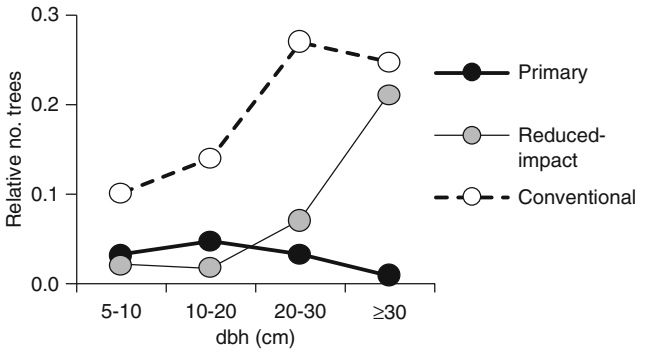


Fig. 3.7 Number of trees of pioneer species relative to the total number of trees by size class in the three 2-ha plots

some species in common with the conventionally logged forest in canopy and understory layers. The primary forest was dominated by dipterocarp species with low abundance of pioneer species (Table 3.1; Fig. 3.5a,b). In contrast, the conventionally logged forest was dominated by pioneer species (mostly *Macaranga* spp.). The reduced-impact logged forest demonstrated a somewhat intermediate pattern; it was dominated by dipterocarp species but with pioneers (also mostly *Macaranga* spp.) accounting for more than one-fifth of the stems ≥30 cm dbh, as seen in the conventionally logged forest (Fig. 3.7). Prevalence of large pioneers in the two logged forests suggests that the pioneer species became established in greater numbers soon after logging, which is the factor explaining the similarity of species composition between the two logged forests in the canopy layer.

The primary forest had similar species composition between small- and large-sized trees within the forest (Fig. 3.6), indicating a continuous regeneration of climax species such as dipterocarp species. The conventionally logged forest had a different composition between small- and large-sized trees, and the reduced-impact logged forest showed an intermediate pattern between the two forests (Fig. 3.6). This finding indicates that the regeneration of climax species becomes disrupted as logging intensity increases, probably because of the decreased seed supply, modified habitat condition, or competition with pioneer species recruited following the logging.

At the small size class (5–20 cm dbh), species richness decreased with increasing logging intensity for Moraceae (9, 4, and 3 species in the primary, the reduced-impact logged, and the conventionally logged forest, respectively), Ebenaceae (13, 4, and 7 species), Anacardiaceae (9, 6, and 4 species), and Dipterocarpaceae (30, 27, and 22 species). Many other families also showed similar patterns.

Reduced-impact logging slightly changed the species composition of small trees (Figs. 3.5e, 3.6b) but maintained similar species composition with that of the primary forest at the medium (10–30 cm dbh) size class (Fig. 3.6b). Medium-sized trees would affect the species composition and diversity of both larger-sized (through their own growth) and smaller-sized trees (through function as a seed source) in the near future. Accordingly, the reduced-impact logged forest may have greater potential to recover to the composition of the primary forest.

3.5 Dipterocarp Regeneration

To understand the regeneration conditions of dipterocarp species, we measured the stem density of seedlings (≥10 cm high and <1 cm dbh) and saplings (1–5 cm dbh) of dipterocarp species in the three 2-ha plots during November 2006–February 2008. We additionally established 100 2 × 2-m plots within the 5 × 5 m plots, which were used to census small trees with 5–10 cm dbh (Fig. 3.1). In the 2 × 2 m plots, each dipterocarp seedling was identified and measured for stem height. In the 5 × 5 m plots, each dipterocarp sapling was identified and measured for stem dbh. In July 2009, all dead, surviving, and recruited (to 10 cm high) seedlings were recorded. Mortality and recruitment rates of dipterocarp seedling were calculated as

$$\text{Mortality rate } (\%\ \text{year}^{-1}) = 100\left(\ln\left(N_o\right) - \ln\left(N_s\right)\right)/t$$
$$\text{Recruitment rate } (\%\ \text{year}^{-1}) = 100\left(\ln\left(N_f\right) - \ln\left(N_s\right)\right)/t$$

where N_o is the number of stems at the beginning, N_s is survived stems, N_f is final stems (N_s + recruits), and t is the census span (years). Some seedlings were difficult to identify at the species level because these were still infertile and often differed from the mature trees in the morphology of vegetative characters (leaves, twigs, and bark). Therefore, such seedlings were identified at the genus level (i.e., *Anisoptera*, *Dipterocarpus*, *Dryobalanops*, *Hopea*, *Parashora*, *Vatica*, and *Shorea*) and the section level for *Shorea* species (i.e., Brachypterae, Mutica, Ovalis, Pachycarpae, Richetioides, and Shorea).

Because light intensity affects the survival rate of dipterocarp seedlings (Aiba and Nakashizuka 2007) and thus may in turn affect their density, hemispherical fish-eye photographs were taken to assess the light intensity in the center of each 2 × 2 m plot at 1-m height. The photographs were taken under an overcast sky with a Coolpix 990 digital camera equipped with a FC-E8 fish-eye converter (Nikon). Hemispherical fish-eye photographs were processed using HemiView image analysis software (ver. 2.1; Delta-T Devices), which allowed determination of the indirect site factor (ISF), direct site factor (DSF), and global site factor (GSF). ISF and DSF are the fractions of indirect (diffuse) and direct light that penetrates to a particular site relative to the amount of radiation above the forest canopy. GSF is calculated as a weighted average of ISF and DSF. Site factors range from 0 (complete obstruction) to 100 (open sky). We used the GSF as the value of light intensity in this study.

The capacity of the seed supply of dipterocarp species primarily affects the density of dipterocarp seedlings. We calculated the basal area (BA) of each dipterocarp genus for trees ≥10 cm dbh in each of the 20 × 20 m subplots, and considered the generic BA as a factor affecting the density of the congeneric seedlings in the corresponding 2 × 2 m plot (see Fig. 3.1). BA was not determined in the 20 × 20 m subplots located on the borders of each 2-ha plot.

To examine the relative significance of light and seed supply (i.e., GSF and BA of congeneric trees ≥10 cm dbh), we conducted a regression analysis and a model selection using the stepwise Akaike information criterion (AIC). In a default model, the seedling density was regressed against GSF and BA of conspecific trees for each taxon. The model with the lowest AIC was considered to have the best fit for the taxon. We conducted this analysis at the genus level or the section level for *Shorea* species. Differences of the stem densities of dipterocarp species and the GSF among plots were tested by an analysis of variance.

Densities of seedlings, saplings, and small trees were generally greater in the primary and the reduced-impact logged forests than in the conventionally logged forest (Table 3.3). Seedling density (≥10 cm high and <1 cm dbh) in the primary and the reduced-impact logged forests was approximately 29 and 14 times higher than that in the conventionally logged forest, respectively. Seedlings of 10–50 cm height did not occur in the conventionally logged forest. Recruitment rate of dipterocarp seedling was 3.9%, 1.6%, and 0.0% year^{-1} in the primary, the reduced-impact logged, and the conventionally logged forest, respectively. Seedling mortality

Table 3.3 Stem density of dipterocarp species by tree size class and the dynamics of dipterocarp seedlings in the three 2-ha plots

	Primary		Reduced-impact		Conventional	
Density per plot						
Seedling (10–50 cm high;/4 m^2)	2.52^a	± 6.18	0.96^b	± 4.74	–	
Seedling (≥50 cm high, <1 cm dbh;/4 m^2)	1.60^a	± 3.28	1.00^a	± 1.82	0.14^b	± 0.49
Sapling (1–5 cm dbh;/25 m^2)	0.79^b	± 1.34	1.28^a	± 1.49	0.38^b	± 0.86
Small trees (5–10 cm dbh;/25 m^2)	0.55^a	± 0.76	0.38^a	± 0.63	0.16^b	± 0.47
Density (ha^{-1})						
Seedling (10–50 cm high)	6300		2400		0	
Seedling (≥50 cm high, <1 cm dbh)	4000		2500		350	
Sapling (1–5 cm dbh)	316		512		152	
Small trees (5–10 cm dbh)	220		152		64	
Trees 10–60 cm dbh	151		89		90	
Trees ≥60 cm dbh	18		17		5	
Dynamics of seedling (≥10 cm high, <1 cm dbh)						
Recruitment rate (% $year^{-1}$)	3.9		1.6		0.0	
Mortality (% $year^{-1}$)	4.6		1.8		2.6	

Forests sharing the same letters do not differ significantly at $P < 0.05$

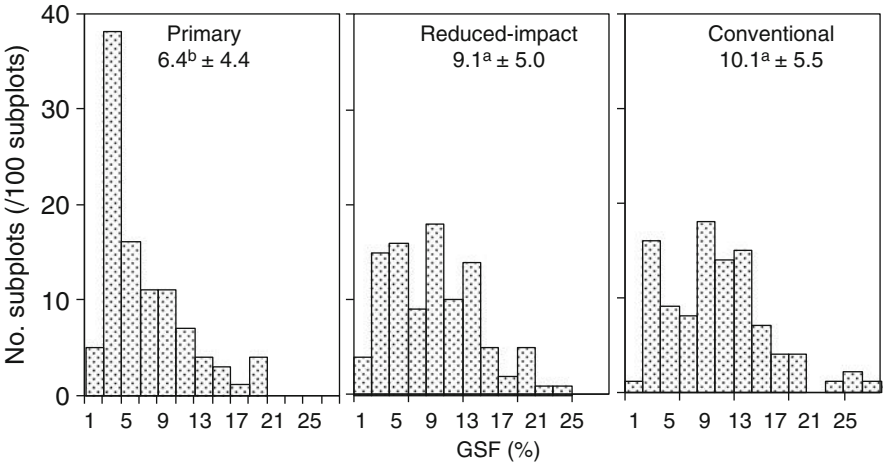


Fig. 3.8 Frequency distribution of global site factor (*GSF*) in the three 2-ha plots. Mean (±SD) GSF is also shown in the figure, and forests sharing the same letters do not differ significantly at $P < 0.05$

tended to be higher in the primary forest (4.6% $year^{-1}$) than in the reduced-impact logged (1.8% $year^{-1}$) and the conventionally logged (2.6% $year^{-1}$) forests.

GSF, which represents the proportion of total radiation available (direct and diffuse) relative to the full sunlight in an open site, was significantly greater in the reduced-impact logged and the conventionally logged forests than in the primary forest (Fig. 3.8). The GSF < 5% (dark condition) appeared in 53 of the 100 subplots

Table 3.4 Regression coefficients and intercepts of the regression analysis for the density of dipterocarp seedlings with basal area (BA) of congeners and global site factor (GSF) in the three 2-ha plots

			Coffecient	
	Genus (section for *Shorea*)	Intercept	BA of congeners	GSF
Primary	*Dipterocarpus*	0.10		
	Hopea	−0.12	**3.39**	**2.40**
	Parashora	0.04	1.20	
	Vatica	0.01	9.00	
	Shorea sect. Brachypterae	0.20		
	Shorea sect. Mutica	0.07	0.25	
	Shorea sect. Pachycarpae	0.23	0.69	−2.21
	Shorea sect. Richetioides	−1.10	**10.88**	18.77
	Shorea sect. Shorea	0.91		
Reduced-impact	*Dipterocarpus*	0.05		
	Dryobalanops	0.02		
	Hopea	0.04		−0.29
	Parashora	0.26		
	Vatica	−0.47	**179.84**	
	Shorea sect. Brachypterae	0.06	**0.72**	
	Shorea sect. Mutica	0.65		−3.24
	Shorea sect. Ovalis	0.01		
	Shorea sect. Richetioides	0.02		
	Shorea sect. Shorea	0.15	0.32	
Conventional	*Shorea* sect. Mutica	0.08		−0.51
	Shorea sect. Richetioides	0.02		
	Shorea sect. Shorea	0.01		

The best-fitting model for each taxon was determined by the stepwise Akaike information criterion

Bold values were significant at $P < 0.05$

in the primary forest, but in only 25 and 22 subplots in the reduced-impact logged and the conventionally logged forest, respectively.

Results of the regression analysis of seedling density are summarized in Table 3.4. Based on model selection by stepwise AIC, BA of congeneric trees ≥10 cm dbh was incorporated into the final model for six of nine taxa in the primary forest, three of ten taxa in the reduced-impact logged forest, and none in the conventionally logged forest, respectively. The direction of the effect was positive for all species, suggesting a positive effect of seed supply on seedling density. Although the GSF also served as an explanatory variable for three of nine taxa in the primary forest, two of ten taxa in the reduced-impact logged forest, and one of three taxa in the conventionally logged forest, respectively, the direction of the effect varied among taxa (positive for two taxa and negative for four taxa) or forest types. In the case of *Hopea*, it was positive in the primary forest but was negative in the reduced-impact logged forest.

Density of dipterocarp seedlings in the reduced-impact logged forest was 14 times higher than that in the conventionally logged forest, and the high density of dipterocarp seedlings in the reduced-impact logged forest may be related to the occurrence of mother trees, but not to greater availability of light. Two logged-over forests (the reduced-impact logged and the conventionally logged forests) showed greater availability of light compared with the primary forest. A brighter condition may contribute to lower mortality in these two logged forests. However, the results of model selection showed that the light effect on seedling density was inconsistent.

BA of large congeneric trees showed a positive effect on seedling density in many dipterocarp taxa. Reduced-impact logging can mitigate the collateral damage to the residual stand, and must leave behind the trees ≥120 cm dbh, which serve as seed sources, according to the RIL guidelines (Lagan et al. 2007). Such practices can contribute to a greater density of large dipterocarp trees, which in turn increases both the density and recruitment rate of dipterocarp seedlings. Although reduced-impact logging yields a lesser amount of timber compared with conventional logging (Putz et al. 2008a), reduced-impact logging can foster future marketable trees.

3.6 Concluding Remarks

The reduced-impact logged forest consistently showed greater α-diversity compared with the conventionally logged forest, irrespective of plot size. On the other hand, the reduced-impact logged forest showed similar α-diversity with the primary forest at a small (0.2-ha) scale, although slightly higher α-diversity than the primary forest at a medium (2-ha) scale. Greater α-diversity at the 2-ha scale is attributable to the greater probability of encountering the patches with high diversity, in which both climax and pioneer species occur, at a larger scale. Accordingly, the reduced-impact logged forest maintained a similar tree diversity with that of the primary forest, although the difference of diversity between the reduced-impact logged and the primary forests varied with the spatial scale of sampling. The reduced-impact logged forest also had a species composition generally similar to that of the primary forest, regardless of plot size. Density of dipterocarp seedlings in the reduced-impact logged forest was 14 times higher than that in the conventionally logged forest because seed supply was secured in the reduced-impact logged forest. Relative intactness in species diversity, species composition, and the density of dipterocarp seedlings in the reduced-impact logged forest is related to the application of reduced-impact logging techniques through reduced harvest and minimized collateral damage to the residual stands. Our results demonstrate that reduced-impact logging is beneficial in maintaining the richness and community assemblage of tree species at a level similar to that of the primary forest if it is practiced strictly following the guidelines developed by the Sabah Forestry Department.

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Chapter 4
Effects of Reduced-Impact Logging on Decomposers in the Deramakot Forest Reserve

Motohiro Hasegawa, Arthur Y.C. Chung, Tomohiro Yoshida, Tsutomu Hattori, Masahiro Sueyoshi, Masamichi T. Ito, and Satoshi Kita

4.1 Introduction

The management of natural tropical production forests is charged to meet with mutually exclusive demands, that is, the production of wood materials and the conservation of biodiversity. One possible way of meeting both demands is to introduce improved management techniques such as reduced-impact logging. The effectiveness of reduced-impact logging in maintaining biodiversity was recognized in selected organisms (e.g., mammals; Mannan et al. 2008). However, this concept needs to be further substantiated in other taxa because benefits of reduced-impact logging can be specific to certain taxonomic groups only.

A forest ecosystem consists of the major functional groups of producers, decomposers, and predators. Classification of biota into such functional groups is beneficial in conservation planning because they are different in tolerance to and resilience from disturbances, response time to disturbances, and overall biological indicator values (Roberge and Angelstam 2009). Here, we introduce the decomposer, one of

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the fundamental groups in the forest ecosystem. Decomposers consist of soil microflora and macro- or mesofauna. Soil microflora play roles in nutrient cycling, plant symbiosis, decomposition, and other ecosystem processes (Nannipieri et al. 2003; Frey et al. 2006). Polypores and other wood-decaying fungi are the primary decomposers of coarse woody debris in forest ecosystems. Soil macrofauna can be divided into two groups: ecosystem engineers and litter transformers (Lavelle et al. 1995). The ecosystem engineers (e.g., earthworms and termites) decompose litter through the action of their internal microorganisms. Therefore, they affect nutrient cycling and soil formation and are important in ecosystem functioning. Litter transformers such as the Isopoda, Diplopoda, and Insecta (e.g., Blattodea and Diptera) decompose litter mainly through the action of external microorganisms. Dipteran fungivorous and detritivores inhabit fallen wood on the forest floor together with fungi, and these organisms, together with their predators or parasitoids, establish a food chain derived from detritus. Collembola and oribatid mites feed mainly on microflora and humus material and are included in micro-food webs (Lavelle et al. 1995).

Decomposers can be indicators of change in forest conditions. Soil microflora have a role as indicators of changes in soil quality because they can respond rapidly to such changes, whereas soil chemical indexes (e.g., carbon or nitrogen concentration) may not directly reflect the soil disturbances resulting from forest management practices (Wardle et al. 1999; Frey et al. 2009). Species composition (including relative abundance and species identity) and richness of wood-decaying fungi are influenced by the quality and quantity of coarse woody debris present in the forest (Bader et al. 1995; Hattori and Lee 2003; Hattori 2005; Gilbert et al. 2008; Snäll and Jonsson 2001). Less-mobile species (e.g., soil macro- or mesofauna) are more useful than mobile species for evaluating the effects of localized logging impacts on biota. Impacts of logging on the forest floor are effectively detected by using soil fauna (Collembola and Oribatida: van Straalen 1997; macrofauna: Doube and Schmidt 1997). Litter-dwelling ants are indicators for evaluating land management and assessing ecosystem conditions because they are sensitive to environmental changes (Underwood and Fisher 2006), including logging disturbance in tropical forests (Belshaw and Bolton 1993; Roth et al. 1994; Vasconcelos 1999; Vasconcelos et al. 2000; Brühl 2001).

Here, we aimed to investigate the effects of reduced-impact logging and conventional selective logging on biodiversity by using decomposers as indicators of disturbance. Selective logging may affect decomposer communities in two ways, namely, changes in physical conditions and resource supply. Both physical conditions and resource supply are changed by high-impact conventional logging, during which soil surfaces become disordered by the introduction of dense trail systems. Such impacts, by contrast, can be mitigated by reduced-impact logging. We therefore anticipate that, under reduced-impact logging, forest community structure and the composition of decomposers are more similar to those in pristine (unlogged) forests than to those impacted by conventional logging. We investigated the effects of reduced-impact logging on various decomposer taxa in the tropical rain forests of the Deramakot Forest Reserve in comparison with conventional logging in the adjacent Tangkulap Forest Reserve. As representatives of decomposers, we selected six taxa: soil microbes, wood-decaying fungi, soil macrofauna (identified to the order level), soil mesofauna (Collembola and oribatid mites), flies (Diptera), and ants.

Because reducing soil damage is often a major emphasis of reduced-impact logging (Putz et al. 2008), monitoring soil-related biodiversity may be a practical way of confirming the validity of reduced-impact logging in protecting soil conditions. We discuss the changes in abundance, species richness, and species composition of decomposers in relationship to physical conditions and resource supply.

4.2 Materials and Methods

4.2.1 Sites

The study sites were the production forests of the Deramakot Forest Reserve and the Tangkulap Forest Reserve in Sabah, Malaysian Borneo (5°14–30′N, 117°11–36′E). Here, we use the term "reserves" for land designated for production. The Deramakot Forest Reserve (551 km^2) is adjacent to the Tangkulap Forest Reserve (275 km^2), and both areas are covered by lowland mixed dipterocarp tropical rain forests (see Chap. 1). At the time of our investigation (2001–2007), the forests in the Deramakot Forest Reserve were being sustainably managed with reduced-impact logging, whereas the forest in the Tangkulap Forest Reserve had already been damaged by conventional logging.

We compared microflora and macro- and mesofauna in a remnant of pristine forest with no signs of logging, a reduced-impact logged forest in the Deramakot Forest Reserve, and a conventionally logged forest in the Tangkulap Forest Reserve. The pristine remnant forest is located within the conservation area of Deramakot and has no record of logging since at least the 1980s. Reduced-impact logging has been used since 1995 in the reduced-impact logged forest. Conventional logging was persistently used in the conventionally logged forest. We established five plots in the pristine forest (2 ha × 1 plot; 0.2 ha × 4 plots), five plots in the reduced-impact logged forest (2 ha × 1; 0.2 ha × 4), and four plots in the conventionally logged forest (2 ha × 1; 0.2 ha × 3). Therefore, plots consist of eleven 0.2-ha plots and three 2-ha plots (see also Chap. 3). We mostly made use of the 0.2-ha plots for decomposer analyses. For all taxa, the eleven 0.2-ha plots were used for sampling. The three 2-ha plots were used only in the study of soil microbes. The abundance and diversity of each taxon were compared among the three logging treatments. Differences in composition among communities were analyzed by multivariate analyses [e.g., principal-component analysis (PCA), and nonmetric multidimensional scaling]. We discuss the responses of each taxon to logging treatment. The details of the methods used to analyze each taxon are described below.

4.2.2 Soil Microbes

We established four survey transects (50 m long) in each of the eleven 0.2-ha plots. We sampled soils at five points along each transect at 10-m intervals and combined the samples into one composite sample. We also divided each of the 2-ha plots into

three topographic positions (i.e., upper, middle, and lower slopes) and sampled soils from nine random points per topographic position, combining the samples to make one composite sample per topographic position. Therefore, we obtained four (in a 0.2-ha plot) or three (in a 2-ha plot) composite samples in each plot.

In general, soil organic matter and soil microbes are concentrated in the soil surface. Surface soils may also suffer from greatest disturbances by logging. For these reasons, we sampled soils from the organic (O) and subsurface mineral (A) horizons in October 2007. The O horizon represented the Oe and Oa horizons, which comprised fragmented litter fractions with an accumulation of fine roots below freshly fallen litter (i.e., the Oi horizon). O horizons were sampled with a circular frame (23 cm in diameter). At the same sampling points below the O horizons, the A horizon (0–5 cm deep) was collected vertically with a 5-cm-long core sampler (diameter, 37 mm).

Samples were taken immediately to the laboratory and were lyophilized for lipid analyses. We used a microbial lipid analysis (extraction of signature lipid biomarkers from the cell membranes and walls of microorganisms) to assess microbial community composition. We used a hybrid lipid extraction method (see Kao-Kniffin and Balser 2007) to extract, purify, and identify phospholipid fatty acids from the cell membranes of microbes in freeze-dried soil samples. Fatty acids were quantified by comparison of peak areas between unknown samples and two internal standards (9:0 nonanoic methyl ester and 19:0 nonadecanoic methyl ester) of known concentration. All lipid analyses were conducted at the Teri Balser Laboratory, Dept. of Soil Science, University of Wisconsin-Madison, USA, courtesy of Dr. Teri Balser.

Lipids cannot represent specific microbial "species" but are more commonly assigned to functional guilds (Kao-Kniffin and Balser 2007). For example, monounsaturated lipids generally indicate gram-negative (Gram−) bacteria, whereas branched-chain lipids generally indicate gram-positive (Gram+) bacteria. The abundance of each functional microbial guild was expressed as the sum of the molar concentrations of the indicator lipids.

4.2.3 Wood-Decaying Fungi

We recorded all coarse woody debris 10 cm or more in diameter (including fallen trees, branches, and dead standing trees), as well as the wood-decaying aphyllophoraceous fungi (including polypores and stereoid fungi) occurring on the debris in each of the eleven 0.2-ha plots. The frequency of occurrence of coarse woody debris ≥10-cm diameter bearing fruiting bodies was counted in each plot for each fungal species. In this analysis, we considered one contiguous woody debris as one count irrespective of its size. For the species occurring on smaller substrates, 16 subplots (5×5 m) were established within each 0.2-ha plot, and wood-decaying fungi were recorded on substrates less than 10 cm in diameter within the subplots. Quantification was made in the same manner as for species on larger substrates, that is, the analysis was based on frequency rather than mass.

4.2.4 Soil Fauna

Soil macrofauna (more than 2 mm in body width) was sampled by hand sorting in five quadrats (each 25 × 25 cm) set at 10-m intervals along a line (40 m) at each of the eleven 0.2-ha plots. The litter layer and topsoil (to a depth of 15 cm) were collected in each sample. Ants were excluded here because they were independently assessed by different methods (see below). For soil mesofauna, a 100-ml soil sample (20 cm^2 × 5 cm deep) including the litter layer was collected with a cylindrical core sampler at each of ten points set at 5-m intervals along another line (45 m) in each 0.2-ha plot. The samples were placed in Tullgren funnels within a few hours after sampling. Soil mesofauna was extracted for 3 days under irradiation with 40 W electric bulbs and preserved in 80% ethanol. Collembola and oribatid mites were identified to species level and counted under a microscope in the laboratory. Sampling was conducted in September to October 2003 and March in 2006.

4.2.5 Flies (Diptera)

We established nineteen 10 × 10-m subplots in each of the eleven 0.2-ha plots and collected coarse woody debris (1.5 m > diameter at both ends ≥ 5 cm, length ≥ 1 m) from 17 to 29 February 2009 in each subplot. The whole length and the diameter at both ends were measured for coarse woody debris as a truncated cone, and then the volume was calculated. Emergence traps were installed on the coarse woody debris in the study plots from 17 February 2009 to 20 October 2009 to collect flies. We selected the three subplots that had the greatest volumes of coarse woody debris among the 19 subplots in each study plot, and placed pieces of debris in an emergence trap at each of the three subplots. An emergence trap consisted of a 1 × 2 m bag of fine netting and a collection bottle. Flies captured by these traps were initially stored in 99% propylene glycol, with the traps emptied every 2 months (Sueyoshi et al. 2009). Specimens were observed under a stereoscopic microscope and identified to family level. Abundance data were standardized to 1 m^3 of coarse woody debris in 0.19 ha.

4.2.6 Ants

We investigated ant communities in the study areas by applying three different sampling methods: bait trapping with canned tuna (2 h), hand-sorting from litter in 50 × 50 cm quadrats on the forest floor, and pitfall trapping (conducted for 2 days). A combination of several sampling methods is preferable for collecting ants that have different characteristics (Bestelmeyer et al. 2000). For example, pitfall trapping can collect only surface-active ants; in contrast, bait trapping can collect ants

easily, but those collected are mainly generalist predators (Bestelmeyer et al. 2000). Hand-sorting on litter can collect those ants that live among the dead twigs and litter, but it tends to miss small ant species.

The ant survey was conducted in 11 plots (four pristine plots, four reduced-impact logged plots, and three conventionally logged plots) during September 2007. Ten samples for each sampling method per plot, giving a total of 330 samples, were collected during the survey. The workers of the collected ants were sorted to the genus level by using the identification keys of Hashimoto (2003), and then to the species level under a microscope (35×).

4.3 Results and Discussion

4.3.1 Soil Microbes

The mean total abundance of lipids (a proxy for microbial biomass) in the O and A horizons (Fig. 4.1) did not differ significantly among the pristine, reduced-impact logged, and conventionally logged forests. The abundance of functional guilds in the O and A horizons in terms of the sum of the molar concentrations of the indicator lipids also did not differ among the pristine, reduced-impact logged, and conventionally logged forests for gram+ bacteria, Actinomycetes, gram− bacteria, saprophytic fungi, ectomycorrhizae, arbuscular mycorrhizae, and protozoa (Fig. 4.2). The molar ratio of fungal to bacterial indicator lipids in the A horizon (Fig. 4.3) in the conventionally logged forest was significantly higher than in the pristine forest ($P < 0.05$) and marginally higher than in the reduced-impact logged forest; also, in the O horizon it was marginally higher in the conventionally logged forest than in both the pristine and reduced-impact logged forests. The molar ratio of Gram+ to Gram− indicator lipids in either horizon did not differ significantly among the pristine, reduced-impact logged, and conventionally logged forests (Fig. 4.3). We used a PCA to determine the differences among microbial community fingerprints by using the molar ratio of indicator lipids to total lipids per guild (Fig. 4.4). PCA of lipid biomarkers suggested that the O horizon and A horizon, respectively, differed in soil microbial community composition among the three forests. The O horizon was associated with the A horizon because the shifts of microbial community composition among the three forests were parallel between the O and A horizons. Logging had a stronger effect on the O horizon than on the A horizon because the conventionally logged plots were placed remotely.

The ratio of fungi to bacteria increased with increasing magnitude of logging intensity and, as a result, the microbial composition in the conventionally logged forest was differentiable from those in the pristine and reduced-impact logged forests. A difference between reduced-impact logged and pristine forests was not evident. The reduced-impact logging operation was thus effective in preserving soil microbial communities.

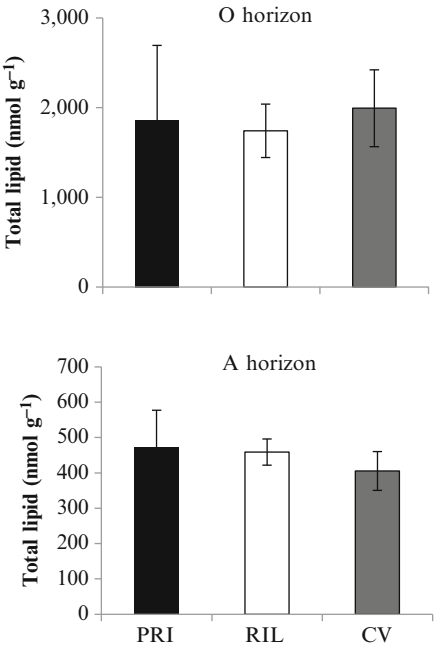


Fig. 4.1 Mean abundances of total lipids in soil samples of O horizon (*upper diagram*) and A horizon (*lower diagram*) in the three forest categories. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest. *Bars* indicate standard deviations of the mean

4.3.2 Wood-Decaying Fungi

We examined the species richness of wood-decaying fungi occurring on coarse woody debris 10 cm or more in diameter (Fig. 4.5) or less than 10 cm in diameter (Fig. 4.6) in each plot. On both the large and the small substrates, species richness in the conventionally logged plots tended to be lower than that in the reduced-impact logged and pristine plots. On the other hand, species richness on both the large and small substrates did not differ between the reduced-impact logged and pristine plots. Recently, in Sarawak, Yamashita et al. (2008) showed that the diversity of wood-decaying fungi declined with increasing intensity of forest use. They suggested that species density of fungi increased with increasing quantity of coarse woody debris. Thus, the lower species richness in the conventionally logged plots may also be partly attributable to the limited quantity of coarse woody debris, because fewer fallen trees are expected to be found in intensively logged forests.

The frequencies of common wood-decaying fungi on the large and small substrates are listed in Tables 4.1 and 4.2. Among the frequently encountered species listed in the tables, all of those recorded in the pristine plots were also found in the reduced-impact logged plots. Moreover, some of the species that appeared in the pristine forest did not occur in the conventionally logged plots. This difference suggests that some common species were likely to be lost in the areas where conventional logging was performed, whereas most species could be maintained in the

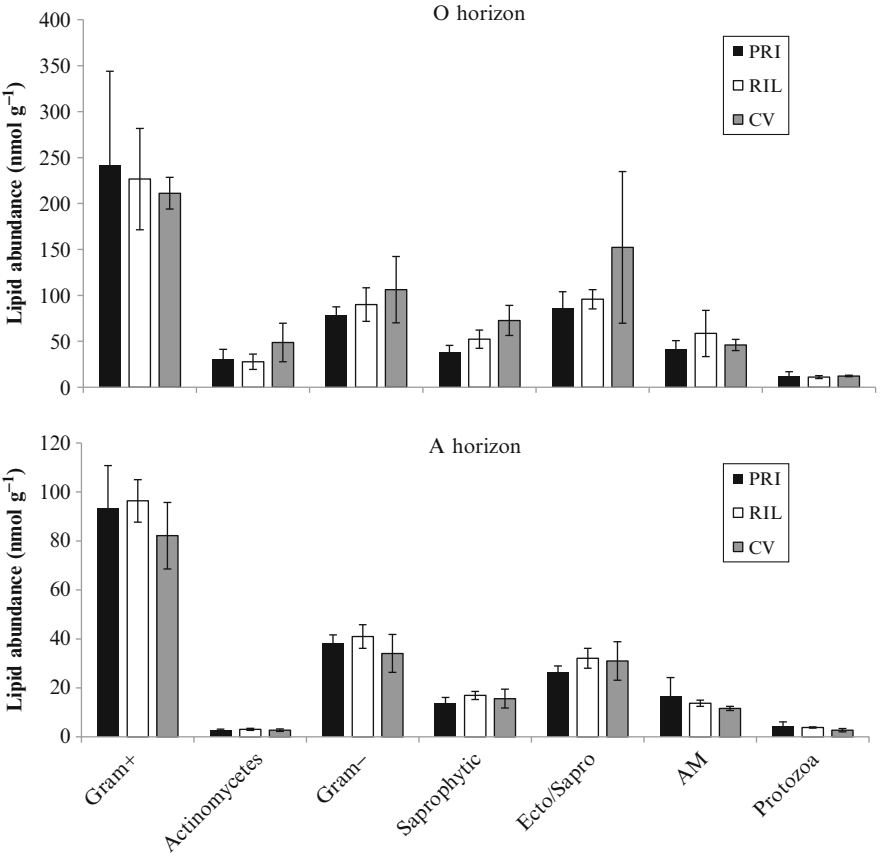


Fig. 4.2 Mean abundance of indicator lipids of each microbial group in soil samples of O horizon (*upper diagram*) and A horizon (*lower diagram*) in the three forest categories. *Gram+*, gram-positive bacteria (iso-branched lipids, the sum of i14:0, 15:0, i15:0, a15:0, i16:0, 17:0, i17:0, and a17:0 lipids); *Actinomycetes*, Actinomycetes (methyl-branched lipids, the sum of 17:0 10Me and 19:0 10Me lipids); *Gram−*, gram-negative bacteria (monounsaturated and cyclopropyl lipids, the sum of 16:1w7, cy17:0, and cy19:9 lipids); *Saprophytic*, saprophytic fungi (the sum of 18:2w6,9 lipids); *Ecto/Sapro*, ectomycorrhizae/saprophytic fungi (18:1w9); *AM*, arbuscular mycorrhizae (16:1w5); and *Protozoa*, protozoa (the sum of 18:3w6,9,12 lipids). *Bars* indicate standard deviations of the mean. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

reduced-impact logging areas. Because of limitations on the area and number of research plots, we were unable to detect the specialists to pristine forests, but some species that were found on trees with massive trunks in the pristine forests were possibly highly specific to those forests. However, many of the species distributed in the pristine forests were also distributed in areas of reduced-impact logging operations; we concluded that the microbiota of wood-decaying fungi was less affected by reduced-impact logging operations than by conventional logging.

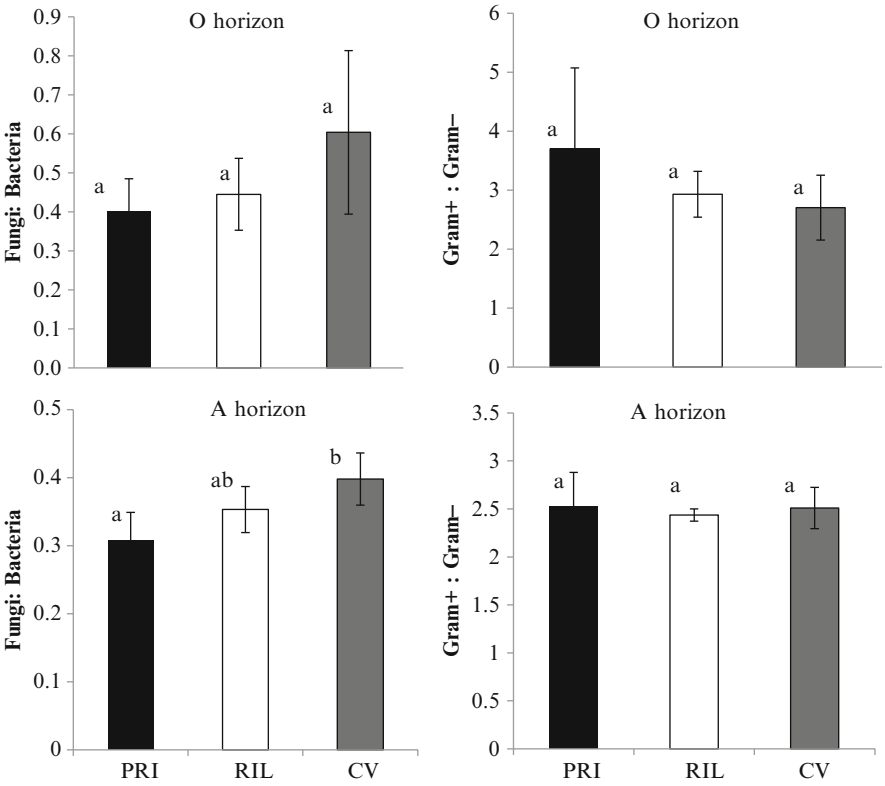


Fig. 4.3 Fungi:bacteria ratio and Gram+:Gram− bacteria ratio in soil samples from the three forest categories. The fungi:bacteria ratio is the ratio of the sum of abundances of fungal lipids (18:2w6,9, 18:1w9) to the sum of all bacterial lipids. The Gram+:Gram− ratio is the ratio of the sum of iso- and methyl-branched lipids to the sum of monounsaturated and cyclopropyl lipids. *Bars* indicate standard deviations of the mean. Results with the same lowercase letters are not significantly different by the Tukey–Kramer HSD. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

4.3.3 Soil Fauna

The abundance of ants was variable among the ten samples in each plot, probably because ant population density depended on distance from the nest. We therefore discuss here the results for macrofauna excluding ants. The mean density of soil macrofauna (excluding ants) did not differ significantly among the three forest categories [nested analysis of variance (ANOVA), $P > 0.05$; Fig. 4.7a]. The number of orders or equivalent taxonomic groups of macrofauna was also similar across the three forest categories (Fig. 4.7b). This tendency was also seen in the density and species richness of Collembola (Fig. 4.8a,b) and oribatid mites (Fig. 4.9a,b), all of which showed little difference among forest categories. Density and species richness of macrofauna and oribatid mites were not significantly correlated with our

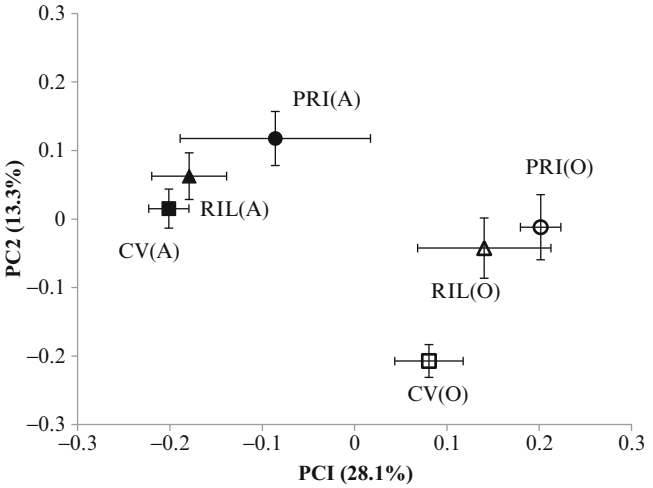


Fig. 4.4 PCA of lipid biomarkers in soil samples from the three forest categories. Lipid abundance was converted to molar percentage values (abundance of each lipid /total lipid abundance). The molar percentage values were not normally distributed. Therefore, the negative arcsine of the square root of each fatty acid molar percentage was used. Each point represents the mean of either A or O horizon of the three forest categories. *Bars* indicate standard deviations of the mean. *PC*, principal component; *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest. (*O*) indicates an O horizon; (*A*) indicates an A horizon

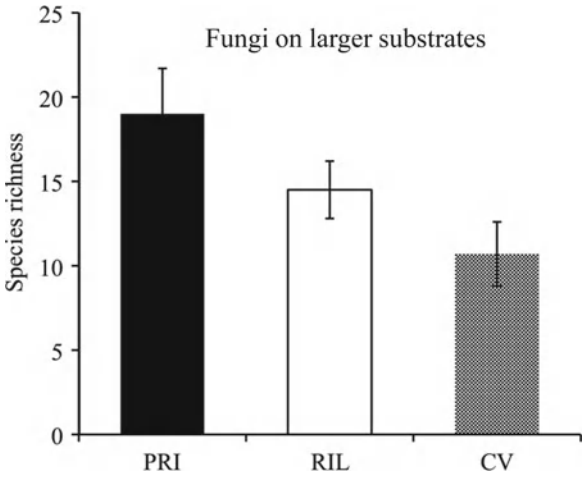


Fig. 4.5 Mean species richness (number) of wood-decaying fungi occurring on coarse woody debris ≥10 cm in diameter in each forest. *Bars* indicate standard deviations of the mean. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

estimated environmental variables (tree density, tree species richness, quantity of forest floor litter, water content of forest floor litter, etc.) (Spearman's rank correlation, $P>0.05$; data not shown). However, species richness of Collembola was significantly positively correlated with the amount of forest floor litter ($P<0.01$, $r_s=0.768$, Spearman's rank correlation; data not shown). The importance of the

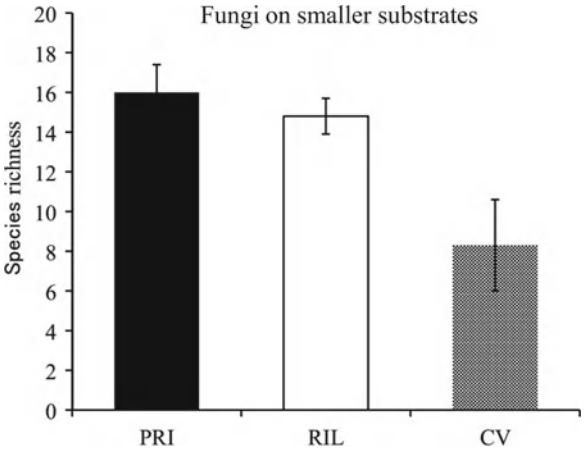


Fig. 4.6 Mean species richness (number) of wood-decaying fungi occurring on coarse woody debris <10 cm in diameter in each forest. *Bars* indicate standard deviations of the mean. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

Table 4.1 Occurrences of wood-decaying fungi recorded frequently on large substrata (≥10 cm in diameter) in the study plots

Forest type	CV			RIL				PRI			
Plot	CV-1	CV-2	CV-3	RIL-1	RIL-2	RIL-3	RIL-4	PRI-1	PRI-2	PRI-3	PRI-4
Earliella scabrosa	3	2				1	1	1			
Erythromyces crocicreas	8	1	5	1	11	1	1	1	1	2	3
Fomitopsis dochmia	5					1	1			3	
Ganoderma australe	4	2	1		1	2	1	2		1	2
Ganoderma mastoporum							1	2	1		3
Hyphodontia ovispora									3	1	2
Microporus affinis	1	2			2	1		3	2	1	4
Perenniporia corticola		1			3	2	1	2	1	2	
Phellinus noxius			2		2	1	1	1			2
Rigidoporus microporus	1	2	1	3	2	1		1			5
Tinctoporellus epimiltinus							1	1		1	2
Trametes menziesii				3				2			1
Trichaptum durum		1				1			1		2

Species with total frequencies ≥5 are listed
RIL, reduced-impact logged forest; *PRI*, pristine forest; *CV*, conventionally logged forest

Table 4.2 Occurrence of wood-decaying fungi recorded frequently on small substrata (<10 cm in diameter) in the study plots

Forest type	CV			RIL				PRI			
Plot	CV-1	CV-2	CV-3	RIL-1	RIL-2	RIL-3	RIL-4	PRI-1	PRI-2	PRI-3	PRI-4
Coriolopsis retropicta	1			5	1	2	1	2		1	
Earliella scabrosa	1	4	3	1							
Erythromyces crocicreas	2							1			3
Hyphodontia ovispora				1		3	2	2	1	1	3
Microporus affinis		1		4	4	3	4	2	1	4	1
Microporus xanthopus	1				1		1	1	4	3	4
Perenniporia ochro-leuca				1			1	1	1		1
Phellinus punctati-formis				1	1	1	2	1	1		
Podoscypha sp. no. 1		2	1					1			4
Polyporus tenuiculus	3	1		1							1
Rigidoporus micropo-rus		4			1		1			1	1

Species with total frequencies ≥5 are listed
RIL, reduced-impact logged forest; *PRI*, pristine forest; *CV*, conventionally logged forest

amount of litter to collembolan community structure has been suggested by Takeda (1987). Accumulation of litter may increase the diversity of collembolan habitat space, which in turn may increase the species richness of Collembola.

Community composition reflected the effects of forest management. Redundancy analysis of macrofauna demonstrated that the water content of the litter layer and species richness of the trees aptly explained the compositional variation of the soil macrofauna community among the plots (Fig. 4.10). The communities in the three conventionally logged plots were thus well differentiated from those in the reduced-impact logged and pristine plots. The structure of collembolan community was well explained by the density of trees (diameter ≥10 cm) (Fig. 4.11). In addition, the structure of the oribatid mite community was well explained by the sum of the basal area of trees (Fig. 4.12). In both results, communities in the conventionally logged plots tended to have compositions different from those in the reduced-impact logged and pristine plots. The water content of litter and soil has been suggested to be an important limiting factor for the survival of some soil animal groups (Lavelle 2001).

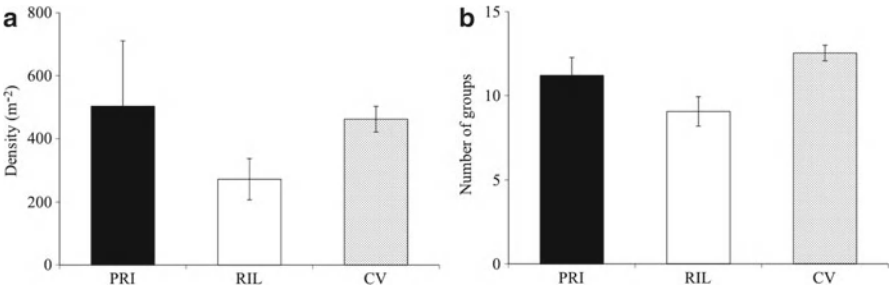


Fig. 4.7 Mean densities (**a**) and mean numbers of taxonomic groups (**b**) of soil macrofauna in samples from the three forest categories, with standard errors. Density of ants was excluded because of high heterogeneity. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

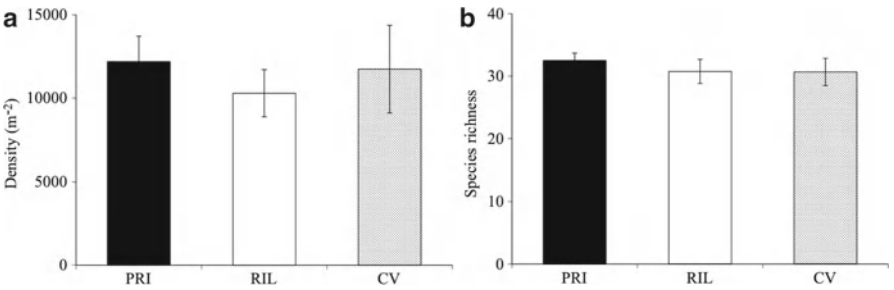


Fig. 4.8 Mean densities (**a**) and mean numbers of species (**b**) of Collembola in soil samples from the three forest categories, with standard errors. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

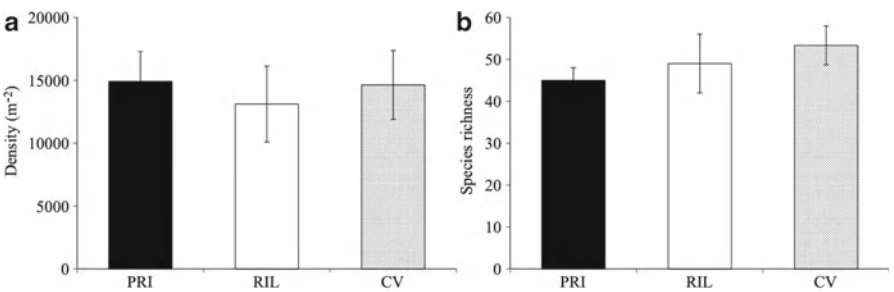


Fig. 4.9 Mean densities (**a**) and mean numbers of groups (**b**) of Oribatida in soil samples from the three forest categories, with standard errors. *PRI*, pristine forest; *RIL*, reduced-impact logged forest; *CV*, conventionally logged forest

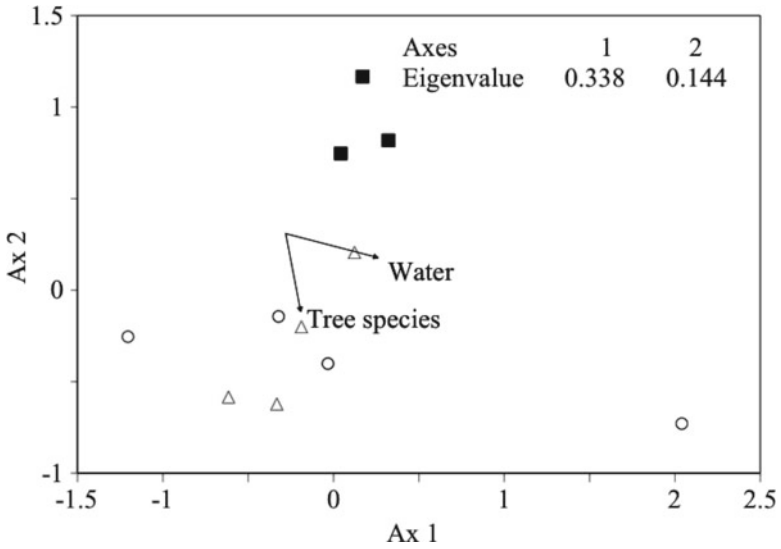


Fig. 4.10 Redundancy analysis ordination diagram for soil macrofauna communities. *Open circles*, pristine forest sites; *open triangles*, reduced-impact logged sites; *closed squares*, conventionally logged forest sites. *Arrows* show directions of the increase in environmental variables, which were extracted by forward selection as significant variables. *Tree species*, species richness of trees; *Water*, water content of litter layer

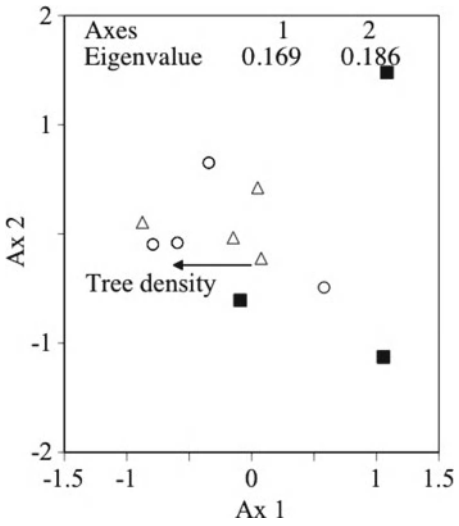


Fig. 4.11 Redundancy analysis ordination diagram for collembolan communities. *Open circles*, pristine forest; *open triangles*, reduced-impact logged forest; *closed squares*, conventionally logged forest. *Arrow* shows the direction of the positive effect of species richness of trees on collembolan communities

On the other hand, tree species richness, tree density, and tree basal area reflect forest maturity (Whittaker 1975). In the process of forest maturation, changes in interrelated environmental factors govern the dynamics of the soil macrofauna community (Scheu and Schulz 1996).

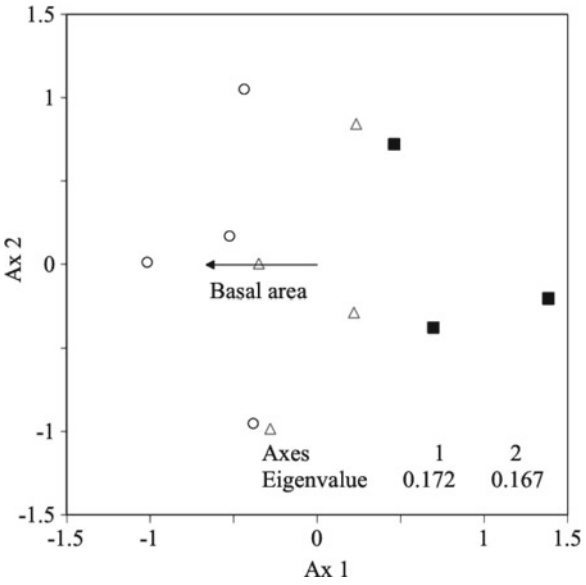


Fig. 4.12 Redundancy analysis ordination diagram for oribatid mite communities. *Open circles*, pristine forest; *open triangles*, reduced-impact logged forest; *closed squares*, conventionally logged forest. *Arrow* shows the direction of the positive effect of tree basal area on mite communities

4.3.4 Flies (Diptera)

The 24 fly families examined (excluding an undetermined family) were divided into three trophic guilds: detritivores (11 families), predators or parasitoids (9 families), and omnivores (4 families) (Table 4.3) (McAlpine et al. 1981, 1987; Papp and Darvas 1997, 1998; Hughes et al. 2000). Flies associated with fungi, rotten wood, and microorganisms were assigned to the detritivores. Predator and parasitoid flies were included in this analysis as factors affecting the abundance of detritivores. Predator flies prey on invertebrates, including detritivores inhabiting decayed wood or those associated with fungi, and parasitoids utilize detritivores as hosts. Omnivore families included both detritivores and predators or parasitoids.

The mean volume of coarse woody debris was 1.24 ± 0.65 m^3 per 0.19 ha in the pristine forest, 0.87 ± 0.20 m^3 per 0.19 ha in the reduced-impact logged forest, and 1.07 ± 0.30 m^3 per 0.19 ha in the conventionally logged forest (Table 4.4). These values did not differ significantly among the three forest types (analysis of variance: $df = 2$, $F = 0.695$, $P = 0.52$).

A principal-components analysis was conducted to ordinate the eleven 0.2-ha plots at family level and to ordinate those families. Figure 4.13a,b demonstrates the same plots of the 11 sample plots, but with predators and parasitoids in Fig. 4.13a and detritivores and omnivores in Fig. 4.13b. The first two (first and second) axes of the principal-components analysis ordination had eigenvalues of 0.350 and 0.226, respectively. The results of PCA based on these two axes demonstrated that the dipteran assemblages were distinct between the pristine and conventionally logged forests (Fig. 4.13a,b). The family assemblage in the older reduced-impact logged plots (RIL3, RIL4: logged in 1995) and younger reduced-impact logged

Table 4.3 Larval feeding habit and abundance of 25 dipteran families in study plots under different types of forest management

Family	Feeding habit	RIL				PRI				CV			N^a
		1	2	3	4	1	2	3	4	1	2	3	
Tipulidae	Om[b]	27.3	10.0	20.3	28.3	70.7	30.3	39.7	17.7	15.0	63.7	11.0	961
Dolichopodidae	Pre[b]	2.3	7.0	19.3	8.0	3.3	10.3	3.7	11.7	6.0	4.0	5.0	225
Sepsidae	Det[b]	0.0	2.7	0.0	0.0	0.0	0.0	2.0	0.0	0.0	24.7	0.0	88
Rhagionidae	Pre[b]	0.0	0.0	12.7	3.0	2.3	1.3	0.0	1.7	0.0	0.7	0.0	65
Milichiidae	Det[b]	0.0	1.0	2.0	12.3	0.7	0.0	1.3	0.0	0.0	0.0	0.5	53
Drosophilidae	Om[b]	0.0	3.3	0.3	1.3	1.0	0.7	2.7	0.3	0.0	1.7	8.0	50
Clusiidae	Det[c]	0.0	1.3	1.7	0.7	0.7	0.0	8.0	0.3	0.0	3.3	0.5	49
Chloropidae	Om[b]	0.0	1.0	0.3	0.7	5.0	0.0	1.3	0.3	0.0	0.7	0.5	29
Empididae	Pre[b]	0.0	1.0	2.0	1.3	1.0	1.7	0.3	1.0	0.0	0.3	1.0	28
Asilidae	Pre[b]	0.7	3.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.3	3.0	20
Muscidae	Om[b]	0.0	0.0	0.3	0.7	1.0	0.3	0.3	1.0	1.0	0.0	1.5	15
Syrphidae	Det[b]	0.0	1.7	0.7	0.7	0.3	0.0	1.3	0.0	0.0	0.0	0.0	14
Micropezidae	Det[c]	0.3	1.0	0.0	0.3	0.0	0.0	0.7	0.0	0.0	2.0	0.0	13
Stratiomyiidae	Det[b]	0.0	0.0	0.3	0.3	0.0	0.3	1.0	0.0	0.0	1.7	0.5	12
Sphaeroceridae	Det[b]	0.0	1.0	1.0	0.0	0.3	0.0	0.7	0.3	0.0	0.0	0.0	10
Calliphoridae	Det[b]	0.0	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	10
Tabanidae	Pre[c]	0.3	0.0	0.3	0.7	0.7	0.3	0.3	0.0	1.0	0.0	0.0	9
Rachiceridae	Pre[b]	0.7	0.0	0.3	0.0	0.3	0.0	0.3	0.0	0.0	0.3	0.0	6
Unidentified family	?	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	6
Lonchaeidae	Pre	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	6
Periscelididae	Det[c]	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	4
Tachinidae	Par[b]	0.7	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	4
Diopsidae	Det[c]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.5	2
Nemestrinidae	Par[c]	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	1
Ephydridae	Det[c]	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	1

Numbers are averages of abundances from three subplots (0.03 ha) per forest type, except for the case of CV1 and CV3. Numbers in CV1 are from a single subplot (0.01 ha), and numbers in CV3 are averages from two subplots (0.02 ha). Abbreviations of feeding habit: *Det*, detritivore (= saprophagy in Hughes et al. 2000); *Om*, omnivore (= mycetophagy, saprophagy, predator, and phytophagy in Hughes et al. 2000); *Pre*, predator; *Par*, parasitoid; ?, unknown

RIL, reduced-impact logged forest; *PRI*, pristine forest; *CV*, conventionally logged forest

[a]Numbers of individuals from all subplots are summed

[b]Larval feeding habit was referenced from Hughes et al. (2000), with some modifications. Here, Empididae and Syrphidae are cited as predator and detritivore, respectively, not as mycetophagy and predator, as in Hughes et al. (2000). Hughes et al. (2000) assigned some of dolichopodids, rhagionids, and stratiomyiids to phytophagous, saprophagous, and predacious insects, respectively. However, these feeding habits were not considered here because they are uncommon in the families

[c]Larval feeding habit was referenced from the work of McAlpine et al. (1981, 1987) and Papp and Darvas (1997, 1998)

plots (RIL1, RIL2: logged in 2000) was not different from that in the pristine and conventionally logged plots, respectively. Dipteran assemblages were distinct among the older and younger reduced-impact logged plots along the second axis. The assemblage in a single conventionally logged or pristine plot was also distinct from those in the other conventionally logged or pristine plots, respectively, along the first axis.

Table 4.4 Volumes of coarse woody debris in the study plots (0.19 ha) under different types of forest management

Plot	Volume of coarse woody debris (m^3)
RIL1	0.84 (0.12)
RIL2	1.05 (0.17)
RIL3	0.60 (0.16)
RIL4	0.99 (0.13)
PR1	2.04 (0.22)
PR2	1.02 (0.15)
PR3	1.41 (0.17)
PR4	0.49 (0.22)
CV1	0.92 (0.16)
CV2	1.41 (0.17)
CV3	0.87 (0.18)

Numbers in parentheses are volumes of coarse woody debris in 0.03 ha, consisting of three 0.01-ha subplots, in which the volume of coarse woody debris was most abundant among the 19 subplots in each study plot
RIL, reduced-impact logged forest; *PRI*, pristine forest; *CV*, conventionally logged forest

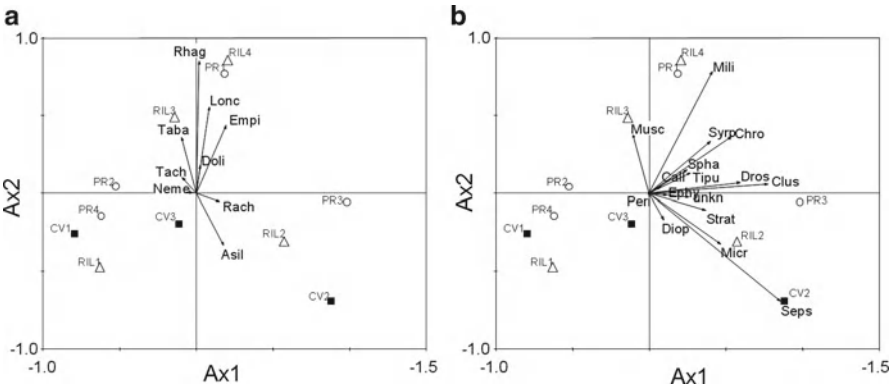


Fig. 4.13 PCA ordination of 25 families of flies (Diptera) and eleven 0.2-ha plots. *Symbols* indicate the positions of the eleven plots from different logging schemes. *Arrows* show directions of increases in abundance for nine predators and parasitoids (**a**) and total 11 detritivores, 4 omnivores, and a family in which feeding habit is unknown (**b**). *RIL*, reduced-impact logging; *PRI*, pristine forest; *CV*, conventional logging. Fly families are indicated with four first letters of each family name indicated in Table 4.3. *Open circles*, pristine forest; *open triangles*, reduced-impact logged forest; *closed squares*, conventionally logged forest

Spearman's rank correlations of the first and second axes of the principal components analysis ordination with the volume of coarse woody debris were 0.70 ($P < 0.05$) and 0.06 ($P > 0.05$), respectively. Therefore, the volume of coarse woody debris changes along the first axis. Forest logging scheme affected dipteran assemblages in at least two ways: one, by affecting the abundance of predacious and parasitoid Diptera in the forest, and the other by changing the physical environment of the forest. The abundance of

Table 4.5 Numbers of ant species collected by bait traps, hand-sorting from litter, and pitfall traps

	Total (11)	PRI (4)	RIL (4)	CV (3)
Bait traps	49	28	23	17
Hand-sorting	104	72	57	45
Pitfall traps	137	74	74	52
Total	205	125	116	87

Numbers in parentheses indicate the numbers of study plots
RIL, reduced-impact logged forest; *PRI*, pristine forest; *CV*, conventionally logged forest

predacious or parasitoid families (excluding Nemestrinidae and Rachiceridae) and detritivorous family of Milichiidae changed along the second axis, which reflected the divergence between the conventionally logged and reduced-impact logged forests (Fig. 4.13a,b). The abundance of 11 detritivorous and 2 omnivorous families, and the predacious family of Rachiceridae, changed positively with the change in volume of coarse woody debris on the forest floor along the first axis (Fig. 4.13a,b).

4.3.5 Ants

A total of 205 ant species (only workers) were recorded by using the three methods (Table 4.5). The observed numbers of ant species in the pristine, reduced-impact logged, and conventionally logged plots were 125, 116, and 87, respectively. Excluding the 33 species found in across all three forest types, the number of species common to the conventionally logged forest and each of the other two forest types (15 and 8) was smaller than the number common to the pristine and reduced-impact-logged forests (34). Ordination of plots using nonmetric multidimensional scaling on a species presence/absence basis revealed no differences ($P>0.05$) in ant community composition between the pristine and reduced-impact logged plots by a multi-response permutation procedure (Mielke 1984), but it did show a significant difference between the former two forests and the conventionally logged forest ($P<0.05$; Fig. 4.14). Our results showed that the forest subjected to reduced-impact logging operations retained ant communities more similar to those of the pristine forest than to those of the conventionally logged forest. This observation suggests that reduced-impact logging is more effective than conventional logging in preserving ground-dwelling ants in Deramakot.

4.4 Conclusion

In Deramakot, the impacts of logging on decomposers are mitigated by the introduction of reduced-impact logging. This tendency was obvious in almost all the taxa that we assessed, but the degree of mitigation effectiveness appeared to differ among taxa. In ants, the practice of reduced-impact logging could maintain species richness at a

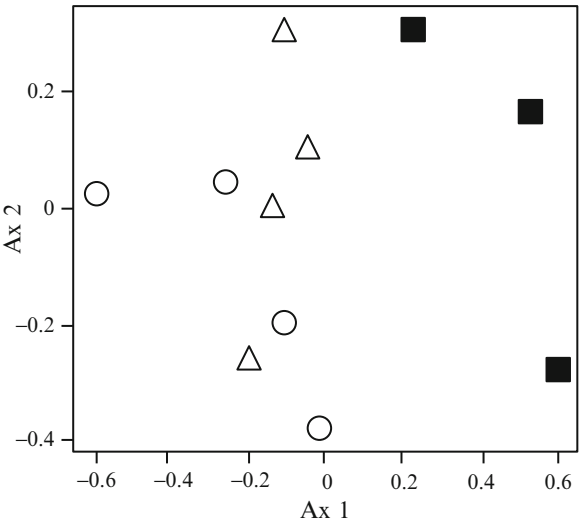


Fig. 4.14 Nonmetrical multidimensional scaling (NMS) ordination analyses of ant communities collected by the three methods (bait traps, hand-sorting, and pitfall traps). *Marks* show positions of the communities in the different forest plots. *Open circles*, pristine forest; *open triangles*, reduced-impact logged forest; *closed squares*, conventionally logged forest

level equivalent to that of the pristine forest. However, species richness of the taxa excluding ants was not sensitive to logging intensity. In all taxa, in contrast, species (or taxonomic) composition was universally similar between communities in the reduced-impact logged and pristine forests, and the communities in the conventionally logged forest were distinct from those in the reduced-impact logged and pristine forests. Comparison of species composition by multivariate analyses clearly showed the effects of logging treatment in terms of the convergence of taxonomic composition between the reduced-impact forest and the pristine forest (Figs. 4.10, 4.11, 4.12, 4.13, and 4.14). This pattern was consistent even in largely cryptic biota including prokaryotes, which is not known to date, because the analysis of soil microbial communities by a biomarker lipid examination demonstrated the same pattern (Fig. 4.4).

The effects of logging on decomposer communities are complex. Logging effects can be transmitted to decomposers through changes in physical condition, food supply, and subsequent biological interactions. Changes in physical condition are related to mechanical perturbations added to soils as well as microclimates associated with shifts in forest structure. For instance, the reduction in crown cover after timber harvest increases the amount of direct sunlight penetrating to the forest floor and thus causes the fluctuation in temperature. Reducing canopy cover allows sunlight to penetrate to the forest floor, where it dries out the organic debris created on top of the soil surface by logging (Nepstad et al. 1999). The changes in temperature may also cause fluctuations in soil moisture content, especially in surface layers. Soil fauna is known to be sensitive to such microclimatic changes as temperature or humidity (Verhoef 1981; Bird and Chatarpaul 1986). Persistent fluctuations in habitat temperature or humidity beyond the resistance/resilience of soil fauna could eventually cause a permanent change in soil fauna community structure. We estimated the water content of litter layers at the time of sampling (data not shown) and found no clear relationship between water content and decomposer density. However, soil or litter water content (and temperature) should be monitored for a longer period,

because such monitoring can reveal a seasonal trend of physical conditions, which may influence decomposer community structure.

Logging can change the quantity and quality of food supply to decomposers. Logging can have mutually confounding effects on the amount and dynamics of wood debris: one effect as leaving behind a great amount of wood debris soon after timber harvest, and the other effect as the removal of above-ground vegetation that can potentially supply wood debris as natural mortality. In our plots, the differences in the quantity of litter (data not shown) or coarse woody debris among the three types of forest are not dramatic. The mode of logging in Deramakot is selective irrespective of either reduced-impact or conventional method, which is in contrast to a clear-cut system, which drastically changes wood debris as shown in Abbott et al. (1980) or Betsch and Cancela da Fonseca (1995) . The relatively small differences in the amount of litter and wood debris (i.e., food source for decomposers) among the three forest types in our case (Table 4.4) may reflect the persistent supply of twigs or logs from non-timber tree species left behind after the harvest or growth of fast-growing pioneer species (e.g., *Macaranga*) at the logging site (see Chap. 3). Similarly, the densities of many decomposer taxa did not differ clearly among the pristine, reduced-impact logged, and conventionally logged forests. We suggest that more or less similar abundances of decomposer taxa among the forests may be related to the invariable food supplies.

Nevertheless, food quality for decomposers must have changed through a shift in litter composition as tree species communities demonstrate significant differences in community composition among the three forests (see Chap. 3). Food quality can influence decomposers through a direct and indirect path. First, a change in food quality will strongly influence substrate-dependent decomposers such as microflora. The host specificity of wood-decaying fungi has frequently been reported (Hattori and Lee 2003; Yamashita et al. 2010). A change in microflora will subsequently be transmitted to a subsequent trophic stage. Most soil fauna feed on microflora and show selective grazing and preferences for given species of fungi (McLean et al. 1996; Newell 1984). Therefore, changes in the microflora community structure could influence the soil fauna community structure. All decomposer taxa in this study showed species (or genera or family) -level compositional differences among treatments. Therefore, changes caused in the tree community composition by logging must have affected the species composition of the decomposer community via resource-quality change in the forests.

We conclude that the number or abundance of decomposer taxa was not sensitive to logging intensity within the magnitude of logging practices in the study area. Taxonomic composition of the decomposer communities was more sensitive to the intensity of logging. If we use taxonomic composition as an indicator of logging impacts, we can conservatively conclude that reduced-impact logging is favorable in maintaining the assemblages of component taxa in decomposer communities. Conventional logging is destructive to the maintenance of decomposer communities and thus to the integrity of forest ecosystems. Responses to logging can be identified well through analyses of species (or family) composition, but not through analyses of the total abundance of decomposer taxa.

Here, we used a multispecies or cross-taxa approach to investigate the effects of logging on biodiversity in a tropical rain forest and demonstrated mutually consistent

results. Logging effects on tree species can spread though a cascade of decomposers. The usefulness of cross-taxon methods has been recognized (Lawton et al. 1998; Gardner et al. 2008). However, when the effects of disturbance on biodiversity are being monitored in cross-taxon surveys in a tropical rain forest, the high cost of biodiversity research becomes a problem (Lawton et al. 1998). We should therefore adopt an objective method for selecting groups of high-performance indicator taxa satisfying both sensitivity to disturbance and cost-effectiveness.

Topic Insect Diversity and Forest Management in the Tropics

By Arthur Y.C. Chung

Ecologically, insects are important in the tropics because of their high diversity. Many plants rely on insects for pollination and dispersal of seeds. Many species in most of the 29 insect orders are involved in pollination. A study by Itioka et al. (2003) has shown that various insect species responded to the flowering in a rain forest in Borneo. Some dipterocarps, for example, are pollinated by thrips (Appanah and Chan 1981).

Some insects are nutrient recyclers and decomposers, such as termites and dung beetles (Jones and Eggleton 2000; Davis 2000). The insects dispose of dead vegetation, animal corpses, and dung. In all these activities, insects process and return vast amounts of nutrients to the soil. Insects are also an important source of food for other insects and for many other animals, such as birds, fish, mammals, amphibians, and reptiles. For example, Lourie and Tompkins (2000) reported that swiftlets feed on arthropods, mainly insects. Lambert (1992) noted that the abundance of insects in the forest might affect the species and populations of insectivorous birds.

Because of their abundance and close relationship with the biotic and abiotic factors in the environment, many insect groups are used in environmental studies as bioindicators. These groups can indicate the effects of habitat changes and fragmentation and the effectiveness of management schemes designed to preserve or change individual species- or community-level patterns. Some of the insect groups that have been studied as bioindicators in Borneo include moths (Chey 2002), dung beetles (Davis 2000), termites (Jones and Eggleton 2000), and soil macrofauna (this volume).

Environmental changes include logging for timber production and conversion of natural habitats for agriculture, human settlement, recreation, amenities, or industry (Meijaard et al. 2005). Although generally insects may not seem to be a crucial group (because they are small, inconspicuous, and abundant) compared with the vertebrates, they should be considered with due respect because of their importance and dominance in the ecosystems. Many insects respond rapidly and dramatically to changes in environmental conditions, making these organisms potentially useful indicators of habitat condition.

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Many of the forested areas were severely logged or converted into plantations. This change has undoubtedly created an impact on insect diversity and ecology because many insects are specific to certain group of plants and are restricted to one kind of habitat. In Brazil, 83% of Coleoptera species are "endemic" to one kind of forest (Erwin 1983).

Lepidoptera are useful bioindicators as they are mainly herbivores and their occurrence will reflect the type of vegetation that constitutes a particular forest area. Chey (2002) compared the moth diversity between lightly and heavily logged sites in the Deramakot Forest Reserve. The heavily logged site gave a higher number of moth individuals, with the proliferation of one particular species, but diversity was higher in the lightly logged site, as measured by the Williams alpha diversity index. The distribution was more even in the lightly logged area and the dissimilarity of the moth fauna between the two sites was high, as measured by Preston's coefficient of faunal resemblance. The higher lepidopteran diversity in the lightly logged areas reflects its species-rich and more equitable community, as diversity incorporates richness, evenness, and rarity of species. Akutsu and Chey (2006) reported that there was no clear difference in flying insect abundance between moderately disturbed plots harvested by reduced-impact logging and undisturbed plots, but the number was distinctly reduced in an intensively logged plot.

In a study on stingless bees by Eltz et al. (2003), also in Deramakot Forest Reserve, the data had shown that potential conflict between timber extraction and bee conservation was reduced when reduced-impact logging guidelines were applied. Harvesting guidelines suggest retaining high proportions of large and hollow trees to preserve stingless bee pollination in sustainable forest management.

The studies highlighted here have generally indicated that logging has adversely affected insect diversity. Thus, to be in compliance with the principles of sustainable forest management, reduced-impact logging harvesting techniques are the way forward. Proper planning before, during, and after logging should be conducted according to the reduced-impact logging guidelines (SFD 2009) to achieve the objectives of the implementation. In addition, monitoring and enforcement would have to be carried out to ensure the effectiveness of reduced-impact logging. In this context, forest law enforcement and governance are crucial. All forest management units in Sabah are required to adopt and practice reduced-impact logging harvesting techniques in their timber extraction and forest management. The cost of the process, however, must be set against any benefit from the conservation of biodiversity. Insects in the forest may benefit from reduced habitat disturbances, but it may prove difficult to persuade logging companies to cut their net profits. Therefore, incentives and recognition, for example, forest certification, would have to be given to forest management units that fully adopt and practice proper reduced-impact logging harvesting techniques.

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Chapter 5
Impacts of Two Different Forest Management Practices on the Abundance of Mammals

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5.1 Introduction

5.1.1 Mammalian Community of Tropical Rain Forest in Southeast Asia

Mammalian faunas of the five tropical rain forest regions of the world (Southeast Asia, Neotropics, Africa, Madagascar, and Australia and New Guinea) have some common characteristics, such as the occurrence of few grazers, high arboreal species richness, and high numbers of frugivore specialists (Whitmore 1998; Primack and Corlett 2005). In general, the body size of mammals in tropical rain forests is relatively smaller than that of the individuals of the same or related species found in more open habitats in the same region (Primack and Corlett 2005). However, there are also differences among the five tropical rain forest regions. For example, Southeast Asia has more diverse sympatric carnivore species (e.g., bears, civets, and cats) than the other regions have (Primack and Corlett 2005, Corlett 2007). The diversities of squirrel and glider species are also high in Southeast Asian forests. On the other hand, the diversity of ungulates and small primates in this region is lower than those in the other tropical regions (Primack and Corlett 2005). Primack and

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Corlett (2005) suggest that the irregular fruiting cycle of the Southeast Asian rain forests is one of the reasons for this pattern. Most of the tree species in a tropical rain forest of Southeast Asia flower and fruit synchronously only at intervals of several years, consequently limiting food resource for frugivores (Primack and Corlett 2005; Corlett 2009).

5.1.2 Logging Impacts on Mammalian Communities and Their Mitigation Through Improved Management Practices

Although tropical rain forests have a high biodiversity and thus have a high conservation value (Richards 1952; Whitmore 1998), most of the remaining tropical forests are designated for commercial timber production (Dennis et al. 2008). Direct and indirect impacts of logging activity on biodiversity have been widely studied (Burgess 1971; Heydon and Bulloh 1996; Whitman et al. 1998; Willott et al. 2000; Fimbel et al. 2001; Laurance 2001; Costa et al. 2002; St-Laurent et al. 2007; Dennis et al. 2008; Corlett 2009). Logging impacts on medium to large mammals in particular have received much attention because of their charismatic popularity (Laurance 2001; Wells et al. 2007). For the common mammal species of Borneo, Meijaard et al. (2005) reviewed many studies that investigated logging impacts on their abundances.

Because conventional logging is known to cause many deleterious social and environmental impacts, various mitigation measures in logging practices have been recommended (Marcot et al. 2001; Mason and Putz 2001; Meijaard et al. 2006; IUCN 2007). Sustainable forest management (SFM) is an integrated management concept that aims at making a profit from timber production while maintaining environmental and social services (Cerutti et al. 2006; Dennis et al. 2008). Minimizing the negative impact on biodiversity is an important criterion of sustainable forest management. Although SFM is implemented both in natural forests and plantation forests, that of natural forests is expected to contribute substantially to the conservation of biological species.

The standards of the SFM of natural forests have been developed by the Forest Stewardship Council (FSC) (Forest Stewardship Council 1996; Vogt et al. 1999). It is a series of management processes: (1) allocating an adequate area for conservation, (2) calculating annual allowable cutting (AAC) based on data of standing stock and prediction of future growth of harvestable tree species, and (3) adopting reduced-impact logging (RIL) techniques. In RIL, all targeted trees to be harvested must be selected and mapped in advance, and an efficient layout of skid trails to minimize impacts is designed before the entry of tractors. Target trees are cut down with a directional felling technique. These measures are considered to be effective to mitigate logging impacts at first hand. However, the criteria of FSC further require monitoring the post-performance of RIL (Forest Stewardship Council 1996). The results of monitoring can feed back to the management system to improve its effectiveness and efficiency.

However, not many studies have thoroughly investigated the impacts of SFM on biodiversity (Dennis et al. 2008). In this study, we investigated the performance of SFM by comparing the abundance and species composition of ground-dwelling mammals and birds among three forests of different management histories: pre-harvest and post-harvest areas in a forest management unit under a SFM scheme (Deramakot Forest Reserve), and the area that was harvested by conventional logging (Tangkulap Forest Reserve).

5.2 Materials and Methods

5.2.1 Mean Trapping Rate Generated by a Random Camera-Trapping System

We placed 29 plots in the Deramakot Forest Reserve and the Tangkulap Forest Reserve in a systematic manner in grids of about 5 km (Fig. 5.1). All forests where the plots were placed were logged at least once by the 1980s. Among these, 15 plots were placed in the portion of Deramakot that had not been logged since 1988 ("pre-RIL forest"). Five plots were placed in the portion of Deramakot that was logged again in 1995–2008 but with RIL techniques ("post-RIL forest"). Nine plots were placed in Tangkulap, which was logged again from the 1990s to 2002 with a conventional logging system ("Conventionally logged forest").

Each plot was a circle with a 1-km diameter. Within each plot, we randomly selected 12–15 setting points for a camera using statistical software R.2.10.1 (R Development Core Team 2011). We uploaded the latitudes and longitudes of the setting points to a GPS (GPSmap60CSx; Garmin, Osath, KS) and went to the location by motorcycle, boat, or on foot. At each setting point, we installed a camera on a tree about 50 cm above the ground. We used an automatic film camera with a passive infrared sensor (Field Note II; Marifu, Iwakuni, Japan). We first installed three cameras at three setting points in each plot and shifted the cameras to another three points in the plot at 3- to 5-month intervals. Each camera was set up to face open ground, avoiding bushes that intercepted the field of view. A field of view from a camera was 2–7 m^2. Films and batteries were changed every 1–2 months. All plots were assessed nearly simultaneously for 25 months from January 30, 2008 to March 3, 2010.

We selected those setting points in a strictly random manner to allow for heterogeneous environments within a plot. If setting points are selected arbitrarily, detection probability (mean trapping rate per true density of a species) can vary erroneously among plots because the difference in environments among plots may result in a skew in detecting the abundance of mammals per plot. For example, if we set cameras only on old skid trails, the detection probability of the animal species that prefer skid trails may increase, and abundance can be overestimated in the plots where the density of skid trails is low because the animal species concentrate on trails. Random selection is also necessary to avoid investigators' personal preferences and biases.

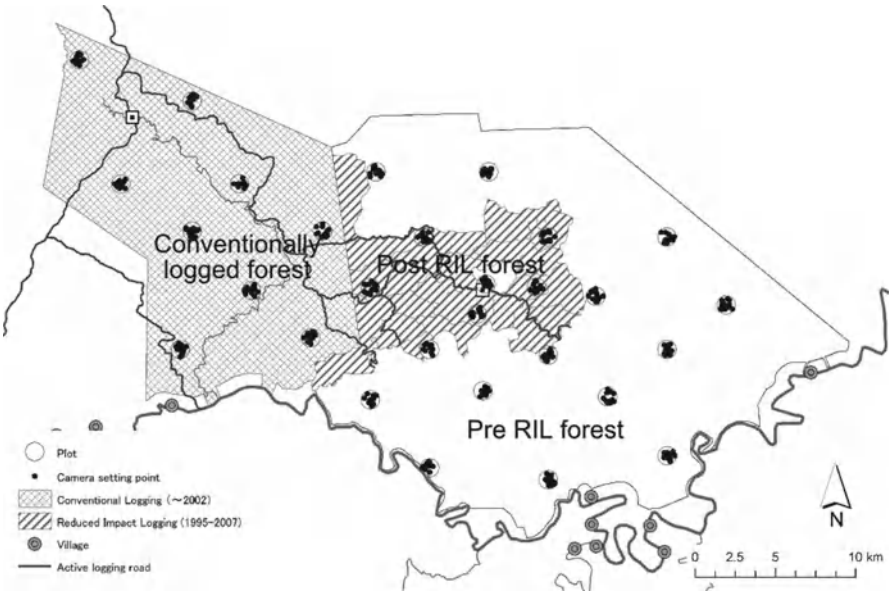


Fig. 5.1 Twenty-nine plots established in three areas with different management histories in the Deramakot Forest Reserve and the Tangkulap Forest Reserve. The conventionally logged forest is *shaded*, the post-RIL forest is *diagonally shaded*, and the pre-RIL forest is indicated without *shading*. *RIL*, reduced-impact logging

Most of our research areas are covered by logged-over secondary forests with a dense understory. As a result, many setting points were not easy to reach. Because there were no existing trails to reach nearly all setting points, we had to make our trails first. Camping was necessary for plots far from the main roads in the beginning but with time we could visit all setting points within a day.

After we retrieved films from the field and developed the photographs, we calculated "mean trapping rate" for each species in every plot. Mean trapping rate of a plot is defined as the number of total photographs of a given species per total active period of all points inside the plot (a 1-km-diameter circle). An "active period" of each setting point is the period from the date when a camera was set at that point until the date when the camera was removed. The periods, during which a camera was not active (i.e., ran out of film or had mechanical malfunction) were excluded from the "active periods." We further excluded redundant photographs when the same species was photographed within 30 min.

Photographs were assessed for all animal species larger than *Tarsius bancanus* and *Echinosorex gymnurus*. We excluded all rats, squirrels, tree shrews, and bats because of the difficulties of identification. Because *Tragulus napu* and *T. javanicus* were often hard to distinguish from each other, we treated these species as one morphospecies. Bornean yellow muntjacs (*Muntiacus antherodes*) cannot be easily distinguished from Bornean red muntjacs (*Muntiacus muntjac*), except for two-spikes

antlers that are the sign of male Bornean red muntjacs. We considered all muntjacs to be Bornean yellow muntjac (*M. antherodes*) because we did not find any two-spikes antlers in photographs. There is also the possibility that what we have recognized as a single species of monitor lizard and a single species of small clawed otter may actually include several species, respectively.

Species richness per plot (1-km-diameter circle) must be compared among plots as the number of species photographed during the same length of active period for all plots. Because the length of total active days was different among plots, we randomly selected 980 camera days during the active period in each plot and counted the number of species photographed during that time. We reiterated random selection for 500 times in each plot, and calculated the mean species number.

5.2.2 Statistical Analysis

We used the mean trapping rate of each species in each plot as the index of abundance and tested differences of mean trapping rate of each species and species richness among the three forests of different management histories by a Kruskal–Wallis rank-sum test. We also tested the difference between any two forests by a Mann–Whitney test.

We evaluated similarities of species composition among the plots by a nonmetric multidimensional scaling (NMDS). The similarities between plots were calculated as Jaccard distance, and differences of the similarity were tested by a Mantel test between (1) pre-RIL forest and post-RIL forest and (2) pre-RIL forest and conventionally logged forest.

5.3 Results

A total of 11,282 photographs of 36 mammal species (35 morpho-species), three ground-dwelling bird species, and one monitor lizard were captured during a total of 34,386 camera-days (Table 5.1). The total active period per plot ranged from 983 to 1,762 camera-days. Of the species captured, seven species were endangered and 10 were vulnerable species in the IUCN Red List of threatened species 2009 (IUCN 2009).

5.3.1 Difference of Mean Trapping Rates and Species Richness

Among the 39 species, mean trapping rates of six species were significantly different among the three forests with different management histories (Kruskal–Wallis test, $P<0.05$; Fig. 5.2). These species are ground-dwelling civets (Malay civet and

Table 5.1 Mean ± standard deviation of trapping rate for 39 species in Deramakot and Tangkulap

					Deramakot				Significance (Mann–Whitney test)		
Common name	Species	Arboreal	Threatened status	Main food habit	Pre-RIL	Post-RIL	Conventionally logged forest (Tangkulap)	Significance (KW test)	Pre-RIL–Post-RIL	Pre-RIL–Conventional	Post-RIL–Conventional
Lesser mouse deer and greater mouse deer	*Tragulus napu and T. kanchil*			HF	712.38 ± 546.83	742.53 ± 185.03	599.05 ± 221.64				
Bearded pig	*Sus barbatus*		VU	O	195.73 ± 162.55	213.81 ± 83.55	332.03 ± 188.83				
Pig-tailed macaque	*Macaca nemestrina*	*	VU	O	223.40 ± 141.19	262.19 ± 130.40	228.27 ± 155.72				
Great argus	*Argusianus argus grayi*	*		O	278.40 ± 176.12	237.36 ± 168.36	107.91 ± 95.16	*		*	
Bornean yellow muntjac	*Muntiacus atherodes*			HF	294.44 ± 186.28	286.13 ± 137.85	49.86 ± 54.25	**		***	**
Long-tailed porcupine	*Trichis fasciculate*			HF	194.75 ± 140.56	264.64 ± 166.79	216.85 ± 217.92				
Moonrat	*Echinosorex gymnurus*			I	157.51 ± 142.18	134.72 ± 123.88	147.44 ± 105.89				
Malay civet	*Viverra tangalunga*			C	204.35 ± 137.38	160.30 ± 156.22	37.68 ± 25.10	***		***	***
Crested fireback	*Lophura ignita nobilis*			O	158.71 ± 359.11	18.18 ± 18.93	55.98 ± 63.14				

Banded palm civet	*Hemigalus derbyanus*		VU	C	133.60 ± 82.14	114.93 ± 48.22	26.60 ± 47.63	**		***	**
Samber deer	*Rusa unicolor*		VU	HF	109.42 ± 124.25	56.50 ± 33.19	97.91 ± 60.69				
Common porcupine	*Hystrix brachyura*			HF	59.37 ± 62.91	27.33 ± 32.53	91.44 ± 71.72				
Malay badger	*Mydaus javanensis*			C	79.06 ± 62.03	28.11 ± 22.03	52.34 ± 49.29		*		
Chestnut-necklaced partridge	*Arborophila charltonii*			O	54.04 ± 41.77	19.93 ± 23.72	23.89 ± 44.45	*		*	
Sun bear	*Helarctos malayanus*		VU	O	34.47 ± 25.35	63.16 ± 16.47	28.21 ± 23.83		*		*
Orangutan	*Pongo pygmaeus*	*	EN	HF	18.92 ± 15.13	13.49 ± 17.49	33.90 ± 27.98				
Monitor lizard	*Varanus salvator*			C	25.37 ± 29.45	10.30 ± 3.78	17.52 ± 14.47				
Thick-spined porcupine	*Thecurus crassispinis*			HF	26.37 ± 35.20	0.00	20.83 ± 29.68				
Short-tailed mongoose	*Herpestes brachyurus*			C	24.79 ± 14.47	26.03 ± 20.65	8.82 ± 9.47	*		*	
Common palm civet	*Paradoxurus hermaphroditus*	*		O	27.01 ± 38.48	13.87 ± 13.06	5.62 ± 9.28			*	
Pangolin	*Manis javanica*		EN	I	11.96 ± 12.88	8.65 ± 6.24	13.60 ± 11.03				
Tembadau	*Bos javanicus*		EN	HF	18.50 ± 48.76	0.00	6.32 ± 10.86				

(continued)

Table 5.1 (continued)

Common name	Species	Arboreal	Threatened status	Main food habit	Deramakot		Conventionally logged forest (Tangkulap)	Significance (KW test)	Significance (Mann–Whitney test)		
					Pre-RIL	Post-RIL			Pre-RIL–Post-RIL	Pre-RIL–Conventional	Post-RIL–Conventional
Western tarsier	*Tarsius bancanus*	*	VU	I	7.57 ± 10.67	3.12 ± 6.98	20.52 ± 16.66	*		*	
Borneo elephant	*Elephas maximus*		EN	HF	8.04 ± 17.07	0.00	15.74 ± 21.23				
Yellow-throated marten	*Martes flavigula*	*		C	10.59 ± 9.92	6.97 ± 11.45	3.88 ± 4.64				
Clouded leopard	*Neofelis diardi*	*	VU	C	6.16 ± 6.43	5.12 ± 11.46	3.83 ± 4.57				
Banded linsang	*Prionodon linsang*	*		C	5.85 ± 11.53	0.00	6.27 ± 7.77				
Otter	*Lutra spp. & Aonyx cinerea*		VU	C	3.83 ± 6.21	0.00	8.82 ± 10.19				
Long-tailed macaque	*Macaca fascicularis*	*		O	3.68 ± 8.95	3.44 ± 4.74	3.61 ± 10.82				
Binturong	*Arctictis binturong*	*	VU	O	3.30 ± 5.01	3.59 ± 4.92	2.73 ± 5.79				
Leopard cat	*Prionailurus bengalensis*			C	2.48 ± 5.59	1.56 ± 3.49	5.06 ± 8.23				
Marbled cat	*Felis marmorata*	*	VU	C	3.99 ± 8.07	1.71 ± 3.82	1.07 ± 3.21				
Otter civet	*Cynogale bennettii*		EN	C	1.07 ± 2.83	0.00	4.72 ± 8.85				

Collared mongoose	*Herpestes semitorquatus*			C	1.27 ± 4.94	5.27 ± 7.70	0.00
Bay cat	*Catopuma badia*		EN	C	1.22 ± 3.33	4.68 ± 10.47	0.00
Flat-headed cat	*Felis planiceps*		EN	C	0.00	1.56 ± 3.49	2.77 ± 5.81
Red leaf monkey	*Presbytis rubicunda*	*		HF	0.63 ± 2.46	1.88 ± 4.20	0.84 ± 2.52
Malay weasel	*Mustela nudipes*			C	0.63 ± 2.46	0.00	0.00
Small-toothed palm civet	*Arctogalidia trivirgata*	*		O	0.56 ± 2.15	0.00	0.00

Shown are the numbers of photos per 10,000 camera-days. "Significance" indicates significant difference in a Kruskal–Wallis test before Bonferroni correlation. "Arboreal" indicates those animals at least partly arboreal (Payne et al. 2005)

EN, endangered species; *VU*, vulnerable species in IUCN Red List 2009 (IUCN 2009)

Main food habit is based on Smythies (1999), Matsubayashi et al. (2007), and Myers (2009): *C*, carnivore; *HF*, herbivore and frugivore; *I*, insectivore; *O*, Omnivore

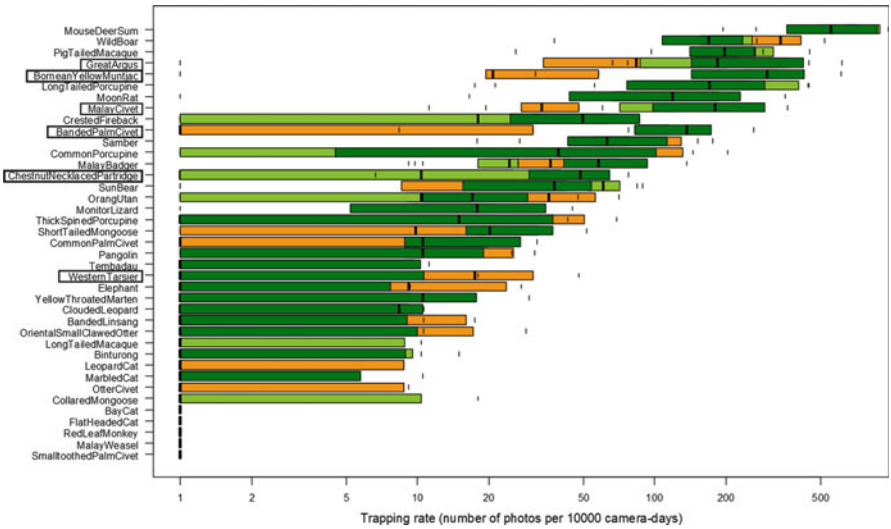


Fig. 5.2 Mean trapping rate of the species photographed in the three forests of different logging histories. The rates are log-transformed after 1 is added. *Green*, pre-RIL; *yellow-green*, post-RIL; *orange*, conventionally logged forest. The species whose mean trapping rates are significantly different among the three areas (Kruskal–Wallis test, $P<0.05$) are shown with a *box around the name*

banded palm civet), pheasants (great argus and chestnut-necklaced partridge), Bornean yellow muntjac, and western tarsier. Mean trapping rates of seven species (great argus, Bornean yellow muntjac, Malay civet, banded palm civet, chestnut-necklaced partridge, short-tailed mongoose, common palm civet) were lower in the conventionally logged forest than in the pre-RIL forest (Table 5.1; Mann–Whitney test, $P<0.05$). Conversely, mean trapping rate of the western tarsier was higher in the conventionally logged forest than in the pre-RIL forest (Table 5.1; Mann–Whitney test, $P<0.05$). In the comparison between pre- and post-RIL forests, mean trapping rate of the Malay badger was higher in the pre-RIL forest (Mann–Whitney test, $P<0.05$), but that of the sun bear was higher in the post-RIL forest. Mean trapping rates of Bornean yellow muntjac, Malay civet, and banded palm civet were significantly higher in the pre- and post-RIL forests than in the conventionally logged forest (Mann–Whitney test, $P<0.05$). Spatial variations of the trapping rates of those notable species are demonstrated in Fig. 5.3.

The species richness observed in 980 camera days per plot was 23.3 ± 2.4 (mean ± SD) in the pre-RIL forest, 20.6 ± 2.2 in the post-RIL forest, and 21.6 ± 1.8 in the conventionally logged forest. However, the differences among the three forests were not statistically significant (Kruskal–Wallis test, $P<0.05$).

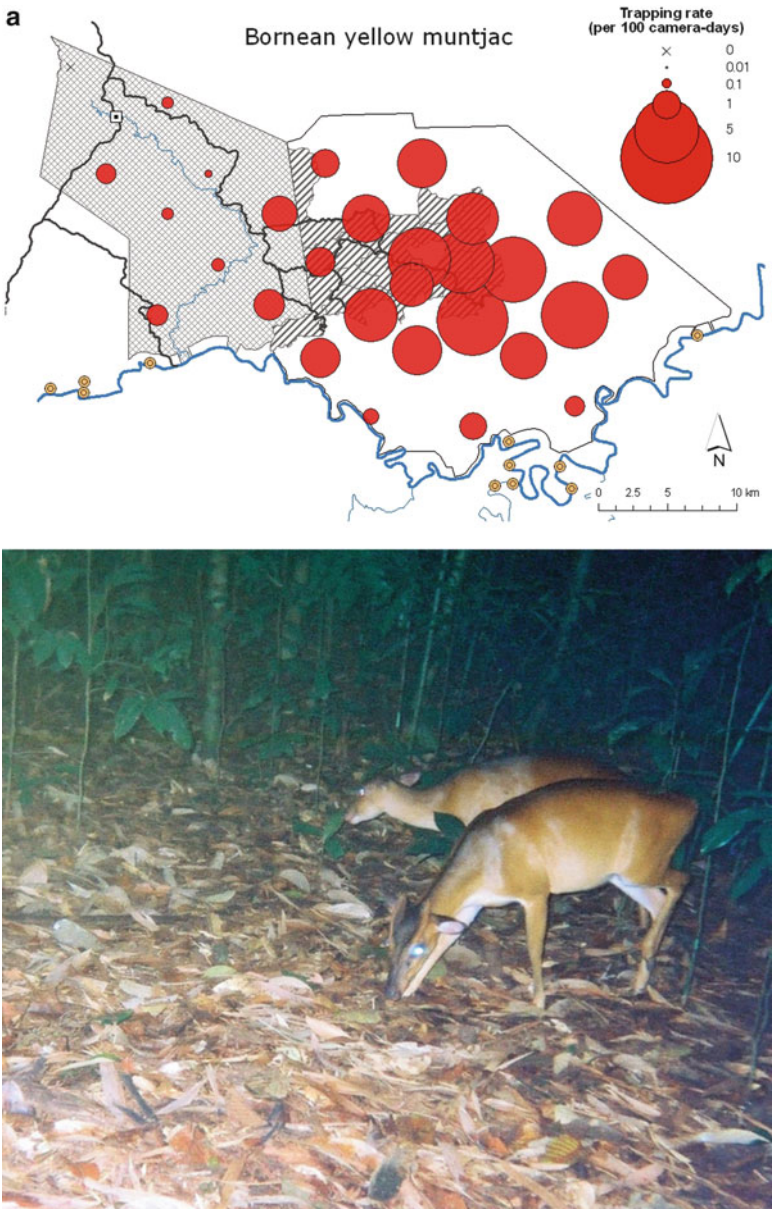


Fig. 5.3 Spatial variation of the trapping rate of several notable species in the study site: Bornean yellow muntjac (**a**); Malay civet (**b**); banded palm civet (**c**); great argus (**d**); chestnut-necklaced partridge (**e**); Malay badger (**f**)

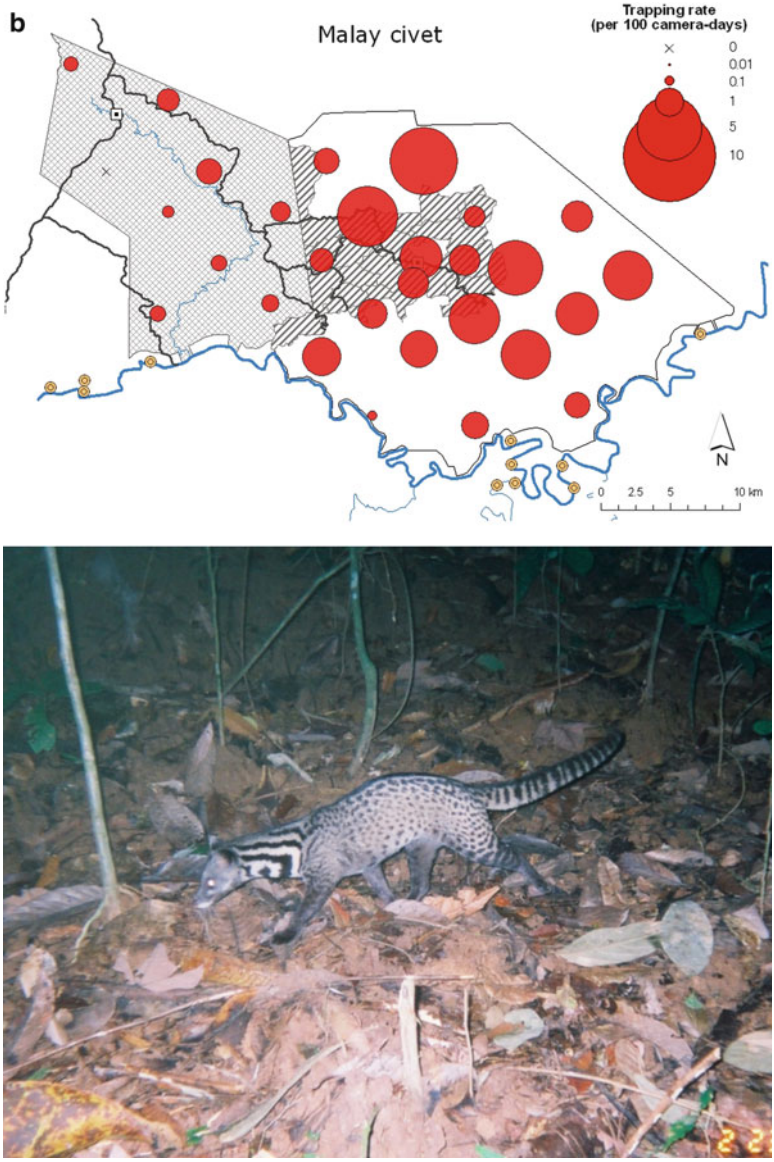


Fig. 5.3 (continued)

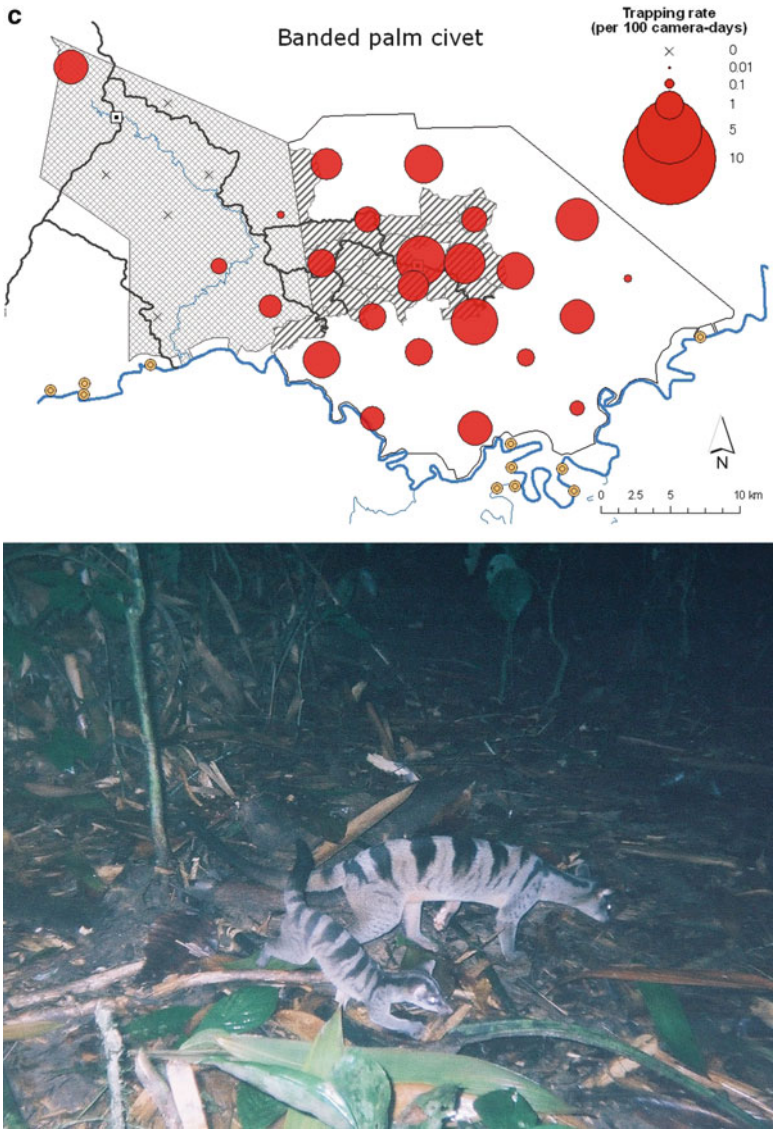


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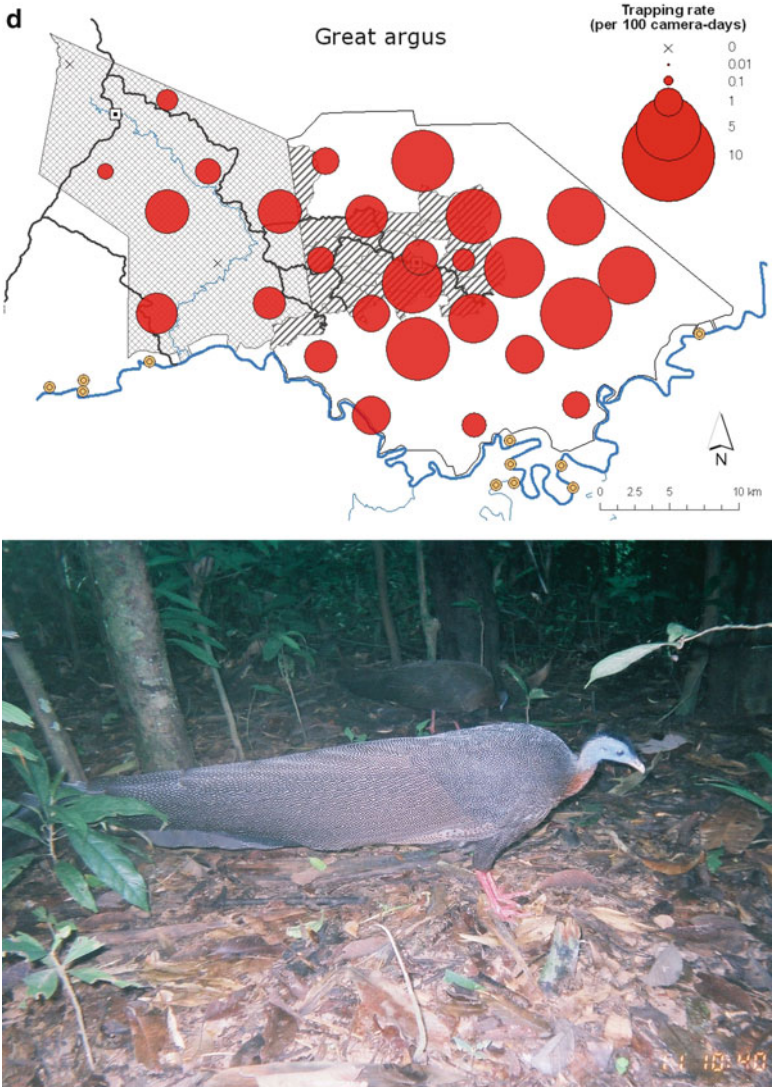


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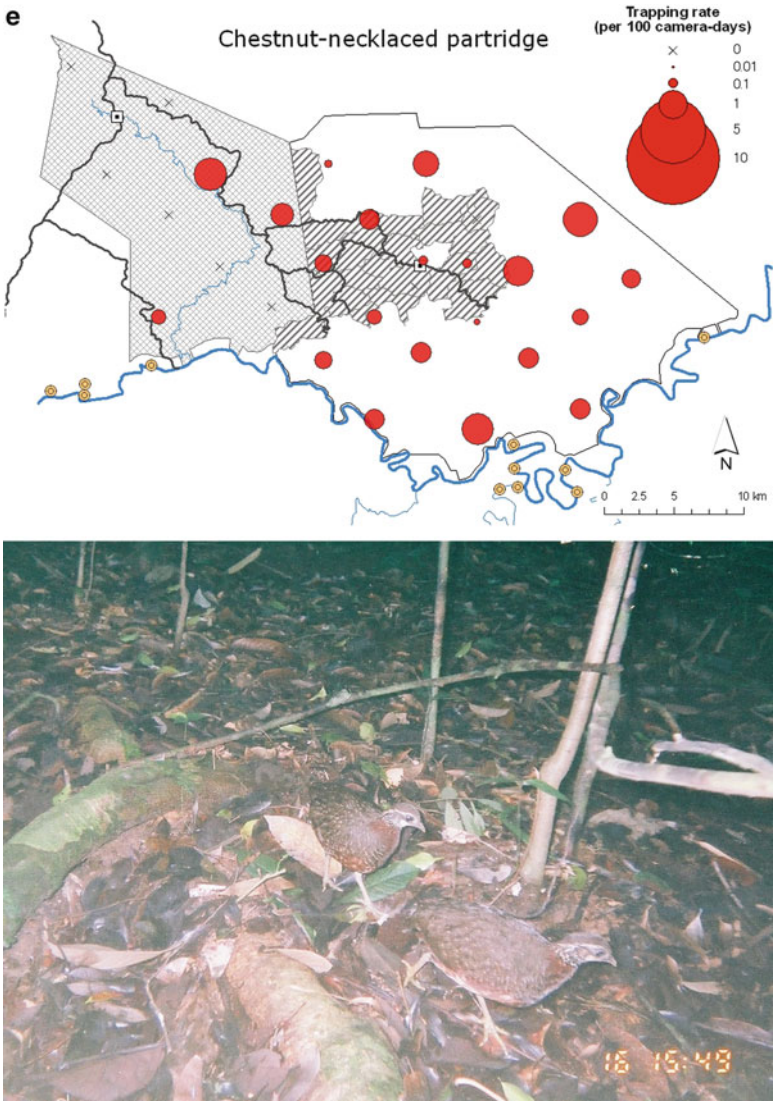


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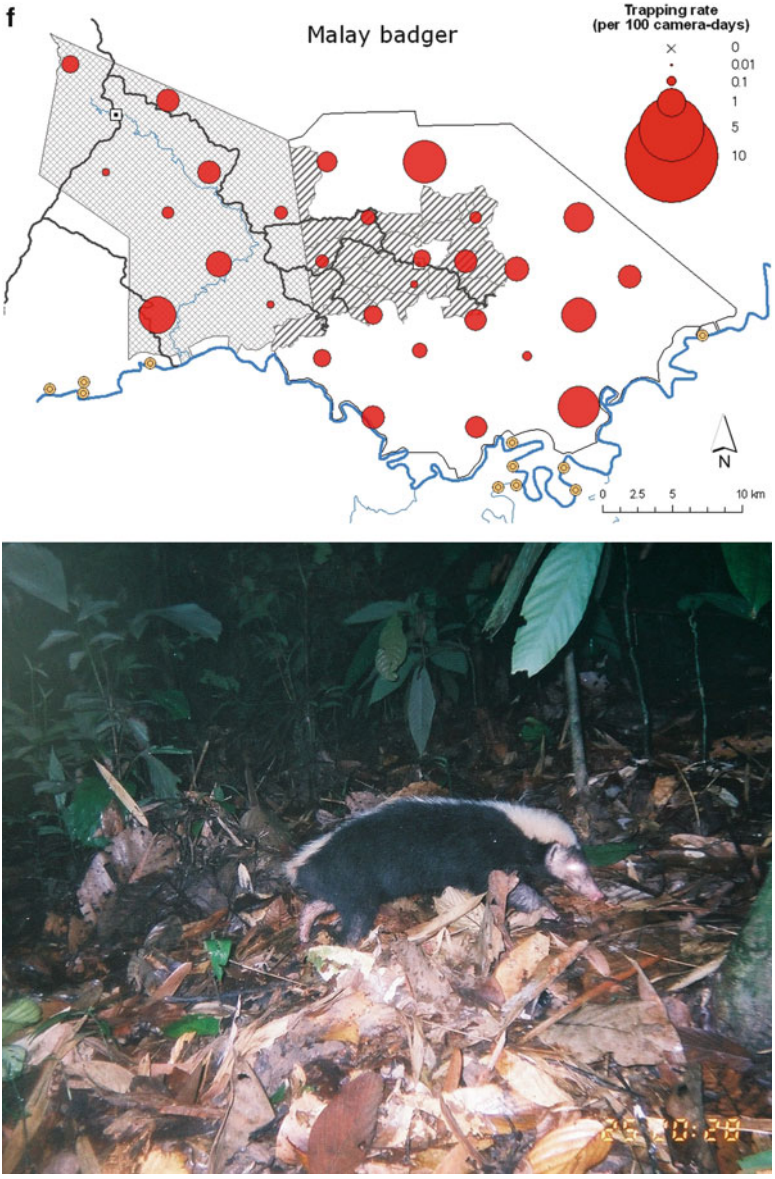


Fig. 5.3 (continued)

Table 5.2 Jaccard distance (mean ± SD) between plots

	Mean ± SD
Pre-RIL–Pre-RIL	0.51 ± 0.22
Post-RIL–Post-RIL	0.32 ± 0.24
Conventional–Conventional	0.45 ± 0.24
Pre-RIL–Post-RIL	0.54 ± 0.1
Pre-RIL–Conventional	0.64 ± 0.08
Post-RIL–Conventional	0.57 ± 0.08

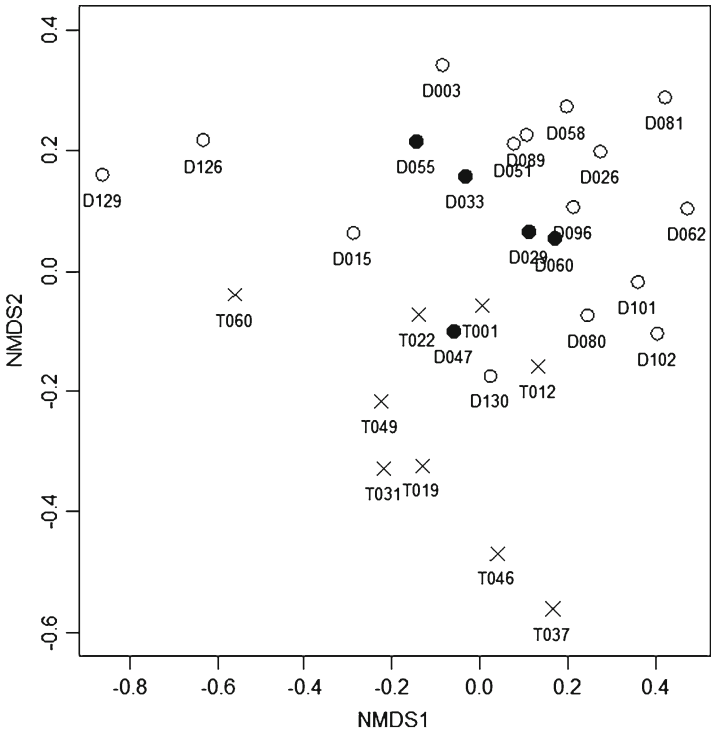


Fig. 5.4 Similarity of species composition among plots shown as nonmetric multidimensional scaling (NMDS): *open circle*, pre-RIL (Deramakot); *filled circle*, post-RIL (Deramakot); *cross*, conventionally logged forest (Tangkulap)

5.3.2 Difference of Species Composition

Figure 5.4 shows the results of NMDS and Table 5.2 shows mean Jaccard distances between the three forests of different logging histories. The difference of species composition between the pre-RIL forest and the conventionally logged forest was significant (Mantel test, $P < 0.01$), whereas the difference between the pre-RIL forest and the post-RIL forest was not significant ($P < 0.05$).

5.4 Discussion

5.4.1 Influence of Logging

Our study demonstrates that the Deramakot and Tangkulap Forest Reserves support a high diversity of mammal species, including many elusive and endangered species. In addition to our records, five mammals (*Cynocephalus variegatus*, *Nycticebus coucang*, *Presbytis cristata*, *Nasalis larvatus*, and *Hylobates muelleri*) have been additionally observed based on camera trapping in Deramakot by H. Samejima (unpublished data). Thus, 41 species of medium-to-large mammals were confirmed to inhabit Deramakot and Tangkulap; they constitute 75.9% of the recorded medium-to-large mammal species for the whole of Borneo (Payne et al. 2005).

The random camera trapping system is a useful method to monitor the abundance of various species and species richness. Meijaard et al. (2005) suggest that the densities of mammals vary considerably between sites even inside an undisturbed forest. Moreover, Karanth et al. (2004) suggest that detection probability (trapping rate per true density) strongly varies among camera setting points. However, we consider that our results are relatively robust because we established 12–15 setting points inside a plot and 5–15 plots in each forest.

Negative impacts of conventional logging on mammalian diversity were obvious. Mean trapping rates of eight species and overall species composition were significantly different between the pre-RIL forest and the conventionally logged forest. Although we do not have data, we can conservatively assume that the abundance of each species studied and species composition were not obviously different before RIL between the pre-RIL forest and the conventionally logged forest because original vegetation and logging history before 1988 were similar. Therefore, the current differences between these two forests are considered to be caused by the impact of conventional logging. These results are consistent with other studies, which also demonstrated significant impacts of conventional logging on the abundance of mammal and bird species (Meijaard et al. 2005). Species richness indicated a different pattern with that of species composition: species richness was not different between the pre-RIL forest and the conventionally logged forest. Bennett and Gumal (2001) also found that logging changed the species composition of mammals in Sarawak but did not change species richness.

Among the species with greater frequencies, ground-dwelling civets (Malay civet and banded palm civet) and short-tailed mongoose were more sensitive to logging than were omnivores and herbivore species (e.g., ungulates and porcupines). The reduction of civets was also noted by Heydon and Bulloh (1996) and Meijaard et al. (2005). However, cats and the clouded leopard did not show an obvious reduction in the conventionally logged forest, even though they can be potentially sensitive. Food resources for civets might have been severely decreased by logging, which may explain the difference in responses among carnivores. It is also notable that, among ungulate species, only the Bornean yellow muntjac was sensitive to logging. One possible reason is that Bornean yellow muntjac is strongly dependent

on fruits and is therefore sensitive to logging (Heydon 1994). However, even though the mousedeer is also known to depend on fruit resources and is sensitive to logging in other study sites of Sabah (Heydon and Bulloh 1997), they did not show a significant reduction from logging in our study site. The sun bear was also decreased by logging in other studies, but it did not show a significant reduction in our study. The main reason for the population reduction in other sites can be increased hunting pressure, which is not the case in our study site. Meijaard et al. (2005) suggests that pheasant is sensitive to hunting pressure but not so sensitive to logging. Because pheasants can fly and are partly arboreal, their correlation between trapping rate and true density may be weak. However, the lower trapping rate of both great argus and chestnut-necklaced partridge in the conventionally logged forest may imply their sensitiveness to logging. Western tarsier is the only one species that is positively affected by logging. As Meijaard et al. (2005) indicated, they are considered relatively tolerant to logging because this species prefers secondary forests.

Our study shows the effectiveness of SFM in mitigating the negative impacts of logging on biodiversity. Mean trapping rates were different between the pre- and the post-RIL forests only for two species. Overall species composition was not significantly different between them. We suggest that the reduction of logging impacts is caused not only by the practice of RIL techniques, but also by the other efforts of SFM including establishing conservation areas and a long cutting cycle. Even though the density of trees to harvest had to be decreased and the skid trails had to be planned to minimize the total length in the comprehensive harvest plan before RIL, the area actually harvested in Deramakot was further decreased because of bad weather and other reasons; only 58.8% of the targeted compartments was harvested. Thus, we assume that most of the ground-dwelling vertebrates must have escaped from the harvested area, at least during the logging operation.

We suggest that the maintenance of the original species composition in Deramakot is explained by three factors: (1) confining the annual harvest to a limited number of compartments supports rapid post-harvest population recovery by migration from the surrounding compartments; (2) RIL prevents the reduction of food and shelter for animal populations; and (3) limited access of hunters to the logging area. However, the trapping rate of Malay badger was significantly lower in the post-RIL forest. For this species, the duration after last logging activity rather than intensity per se may be decisive in determining abundance.

Recently, there has been increasing recognition of the importance of production forests in conserving biodiversity (Putz et al. 2001; Zarin 2004; Ancrenaz et al. 2005; Meijaard and Sheil 2007; Imai et al. 2009; Patterson and Coelho 2009; Edwards et al. 2011). Most medium-to-large vertebrates (except for several species adapting to degraded forests such as *Sus barbatus, Rusa unicolor*, *and Bos javanicus*) need large continuous habitats to maintain viable populations; they may be supported by continuous production forests (although moderately degraded) under sustainable forest management. An isolated, strictly protected area may function to conserve such mammals if the area is large enough to protect the necessary habitats for these species, which is often not the case. The importance of production forests in conservation of medium-to-large vertebrates cannot be overstated.

Even though our study supports the effectiveness of SFM on biodiversity conservation in Deramakot, we suggest that monitoring be conducted in each logging concession to evaluate long-term post-logging impacts, because management systems can differ from each other on the ground even if they are similarly certified as "well managed." One of the reasons for the invariable animal composition between pre- and post-RIL in our study may be related to the scarcity of harvestable trees before logging. Because all our study sites had been harvested at least once by the 1980s, the density of large trees before the latest RIL must have been lower than that of primary pristine forests. Some parts of compartments currently totally lack harvestable trees, and no skid trails enter such areas. As a result, these areas have remained untouched and probably have functioned as a shelter for animals during logging operations. In primary forests where the density of harvestable trees is high, skid trails penetrate with high density and logging impacts may be stronger even if RIL techniques are used. Sist et al. (1998, 2003) indeed detected an insufficient regeneration of canopy species in a logging concession where the density of harvested trees was very high (>10 trees/ha), even though RIL was implemented.

Only eight species were affected even by destructive conventional logging in our study, and mean trapping rates of the other species were not significantly different between the pre-RIL and the conventionally logged forest. This unexpected richness of mammalian fauna in the conventionally logged forest may be related to the networking structure, that is, connectivity with other protected areas. Laurance (2001) showed that even though the abundances of many species was decreased by logging, they could recover in 10 years if the logged forest was connected to unlogged forests. Therefore, connectivity between forest reserves is also important in explaining the extant patterns of mammals. The conventionally logged forest in our study was not an isolated forest, but it was located between Deramakot and a primary, pristine forest reserve. The extended period after logging may be enough to allow many mammal species to return from the surrounding good forests. Therefore, the initial differences between the conventionally logged forest and the post-RIL forest just after logging operation may disappear with time. If an isolated forest reserve is severely harvested, mammal species may not recover.

We compared mammals only between logged-over forests with different logging techniques and different elapsed times after logging. To understand the potential capacity of logged-over forests for the conservation of mammals, we need a further, similar investigation in a totally protected forest reserve. Such intersite comparative studies may help identify those species groups that are truly sensitive to any forms of logging.

5.4.2 Indicator Species to Evaluate Logging Impacts on Biodiversity

Indicator species are useful in evaluating the performance of forest management practices for biodiversity conservation. Those species that have different trapping rates between the pre-RIL forest and the conventionally logged forest in this study

can be considered as species that are sensitive to logging disturbances. Therefore, mean trapping rates of those species and species composition can be used to monitor logging impacts in other areas of similar environmental settings.

The detection probability of partly arboreal species such as pheasants can differ among plots because it may be affected by the density of large trees, which can allow arboreal species to move around. Rare and elusive species have a high conservation value, but they need a very long monitoring period to obtain a sufficient number of photographs for evaluating logging impacts with enough statistical significance. Therefore, they are not a proper indicator species. We suggest that Bornean yellow muntjac and ground-dwelling civet (Malay civet and banded palm civet) be used as a sensitive and legitimate indicator for logging impacts. Continuous monitoring of these species can serve as a means to verify the effectiveness of forest management practices.

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Chapter 6
Guidelines for Establishing Conservation Areas in Sustainable Forest Management: Developing Models to Understand Habitat Suitability for Orangutans

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6.1 Importance of Establishing Conservation Areas for Large Mammals in Tropical Production Forests

Tropical production forests, although degraded, function as refuges for large mammals in Southeast Asia (Alcala et al. 2004; Ancrenaz et al. 2005; Iskandar et al. 2006; Venter et al. 2009) because strictly protected parks are rather limited in area. For example, in Sabah, Malaysia, 47% of total land area remained forested in 2008, of which the majority was logged-over production forests (Sabah Forestry Department 2009). Consequently, more than 60% of orangutans living in the state are found in such degraded production forests (Ancrenaz et al. 2005), clearly suggesting the importance of production forests as a habitat to maintain viable populations of critically endangered orangutans; this is also applicable to other endangered large mammals (see Chap. 5). Therefore, management of modern production forests needs special attention to conserve viable populations of endangered mammals. Conservation strategies need to be incorporated to sustainable management planning, such as establishing conservation areas within or adjacent to the production forests (Lynam et al. 2007).

From another point of view, present-day forestry is required to practice sustainable forest management, that is, to achieve sustainable timber production with a minimum loss of ecosystem services (International Tropical Timber Organization

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(ITTO) 2006). Sustainable forest management in Deramakot (our study site) consists of three components (see Chap. 1): that is, a long-term management plan (which defines the overall cycles of logging, etc.), reduced-impact logging (silvicultural techniques locally applied in a target forest for timber harvest), and forest certification. Forest certification is different from the former two components because this is an institution to verify the sustainable practices and positive environmental, societal, and economic effects on the ground. Although forest certification provides consumers with the assurance of the sustainability of the forests from which the timber has originated, it also provides foresters with incentives for better market access as a result of market mechanisms to drive out uncertified products.

The forest certification system requires compliance with some principles and criteria, which relate to environment, institution, and economy/society. For example, the certification scheme of the Forest Stewardship Council (FSC) requires compliance with ten principles, four of which are directly relevant to biodiversity conservation: principle #6 requires assessment of the impact of forest management on ecosystem functions, including biodiversity, and requires safeguards against this impact, such as conservation areas; principle #7 requires plans for protecting biodiversity and maps describing conservation areas; principle #8 requires monitoring of biodiversity; and principle #9 requires maintenance of high-conservation-value forests (Forest Stewardship Council 1996). Accordingly, all forests certified by the FSC Forest Certification need to comply with those principles to assure biodiversity safeguards. Two of the most efficacious safeguard measures stipulated in the FSC principles to conserve wildlife in production forests are believed to be the establishment of conservation areas and monitoring populations. However, guidelines and procedures for establishing conservation areas for wildlife, particularly for large mammals, have not yet been systematically developed.

A synergy of biodiversity conservation and timber production is often challenging. Productive sites for timber are highly likely to overlap with suitable habitats for large mammals because food resources are abundant to support viable populations of the mammals in such productive sites. Conversely, designating areas that are not productive for timber within a production forest may not be effective for the conservation of large mammals. This conflict often leads forest managers to believe that conservation may not be compatible with economic benefits of timber production. Therefore, designating a conservation area within production forests needs comprehensive discussions among all stakeholders, including the societal supports without which sustainable forest management per se cannot be sustainable.

With these backgrounds, ecologists are charged to provide data on the habitat suitability of a target species of conservation so that measures to synergize timber production and wildlife conservation can be developed. Evaluation of habitat suitability starts from the investigation of the extant distributions and behaviors of a target focal species together with the environments the target species inhabits. In most cases, however, it is difficult to directly observe large mammals in tropical rain forests (whether pristine or degraded) and to accurately evaluate their distributions, because they generally have a low population density and a large home range. In addition, the existing information on the physical and biological condition of the

environment in a landscape context in tropical regions is often limited for reasons of poor accessibility. Even detailed topographic maps are often difficult to obtain. Therefore, environmental explanatory factors for the distribution of mammals need to be simultaneously investigated when surveying the distributions of the mammals per se. In this study, we collected quantitative data of environmental factors in a landscape context by making use of satellite data.

The next consideration is which species to investigate because several mammal species of different behaviors may inhabit the same production forests. When establishing a conservation area, selecting suitable habitats of an umbrella species is an efficient way to conserve a collection of other species, because an umbrella species ranks higher in the food web and has a greater home range that most likely encompasses the home ranges of other species. The Bornean orangutan (*Pongo pygmaeus*) is one of the endangered large mammals (Ancrenaz et al. 2008) and is a typical umbrella species in Southeast Asia. We have selected the orangutan as a target species for conservation and developed a procedure to evaluate suitable habits for this species in Deramakot.

The Bornean orangutan is arboreal and makes a nest (sleeping platform) on a tree top every day. Orangutans make nests by piling up fresh tree shoots, which later turn brown and can be visible from a distance. Ancrenaz et al. (2005) reported that aerial survey of orangutan nests allows accurate estimation of their distribution and population density over a large area. Thus, we also conducted an aerial survey of their nests to elucidate their distributions. Particularly, in this study, we developed a statistical model for evaluating habitat suitability for orangutans by analyzing relationships between the distribution of their nests and environmental factors based on satellite data.

6.2 Methods

6.2.1 Study Site

We conducted this study in the Deramakot Forest Reserve in central Sabah in Borneo (05°15′–28′N, 117°20′–38′E) (Fig. 6.1). Deramakot Forest Reserve has 55,083 hectares (ha) of mixed dipterocarp forests. However, variable logging intensities of past management practices from the 1950s have left the remaining forests in an extremely heterogeneous condition (see Chap. 1; Lagan et al. 2007). Reduced-impact logging was introduced to the Deramakot Forest Reserve as a sustainable management technique in 1995. Subsequently, the reserve was certified as a "well-managed forest" by the FSC in 1997. A large area is in the process of regrowth (see Chap. 2).

The orangutan population in the Deramakot Forest Reserve was estimated at 792 at a density of 1.5/km^2 by an aerial survey of orangutan nests in 2002–2003 (Ancrenaz et al. 2005). Based on this aerial survey, about 11,000 orangutans remain in Sabah, and thus the population in Deramakot represents 7.2% of the

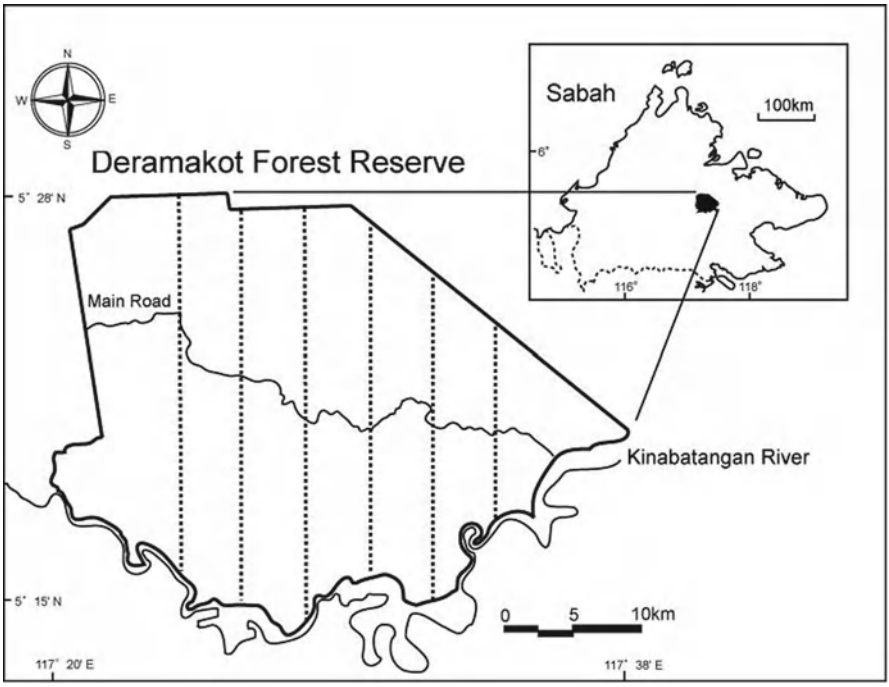


Fig. 6.1 Study area of the aerial census of orangutan nests. *Dotted lines* are the census lines established in this study

total population in Sabah. The population density in Deramakot is the highest among the major orangutan populations in Sabah.

6.2.2 Aerial Census of Orangutan Nests

We established six census transects at even intervals within the Deramakot Forest Reserve. We conducted an aerial census of orangutan nests on these transect lines. Two observers in a helicopter searched for orangutan nests from the air within 30 m on both sides of each transect in November 2007 (Fig. 6.1). The total length of the six transects was 126 km. The observers could spot nests with certainty, as the nests were brown in color and clearly visible on the tree crowns. The geographic position of each nest was recorded by a global positioning system (etrex H, GARMIN) and plotted on a map. Habitat preference of orangutans was judged based on their nest abundances (i.e., nest density). To estimate the spatial pattern of nest density, a total of 673 meshes of 180 m × 180 m were demarcated on the transects, and the number of orangutan nests in each mesh was counted by using GIS software TNT Mips. This mesh size was arbitrarily set to increase the variance of nest densities among meshes.

6.2.3 Data Collection for Environmental Factors

Previous studies suggested that some human-induced factors, such as logging (Johns 1986; Felton et al. 2003; Morrogh-Bernard et al. 2003), proximity to villages known to hunt orangutans (Marshall et al. 2006), and proximity to oil palm plantations (Wich et al. 2008; Nantha and Tisdell 2009), affected the spatial distribution of orangutans in a negative manner. Natural environmental factors, such as fruit availability (Buij et al. 2002) and natural licks (Matsubayashi et al. 2007), could also affect the distribution of orangutans. Therefore, data on the following six anthropogenic factors were collected for each mesh on the basis of satellite data with 5-m resolution (SPOT; July 28, 2003; Scene ID, 5301339030728023552U), maps showing forest management plans (such as the arrangement of roads and compartments), and harvesting plans: distance from the nearest main road that passes through the base camp of Sabah Forestry Department (A), distance from the closest secondary logging road that branches out from the main roads (B), distance from the closest tertiary road, including skid trails (C), logging history, that is, unlogged versus reduced-impact logging (D), distance from the closest oil palm plantation (E), and distance from the closest village around the reserve (F). Roads were classified into three types depending on the traffic conditions and road width, which were indicators of forest discontinuity impeding migration of animals. In addition, four natural environmental factors were also determined for each mesh: distance from the closest natural lick (G), altitude (H), slope inclination (I), and distance from the closest identifiable river on the satellite images (J). The locations of natural licks were determined through interviews with staff members working in the forest reserve. Food availability could not be determined in each mesh, and this factor was omitted in our analysis. For the seven factors, A, B, C, E, F, G, and J, distance from the nearest object to the center of a mesh was calculated using the GIS software. Altitude was calculated from the three-seconds mesh (about 90 m) data of SRTM-3 (Shuttle Radar Topography Mission-3), and slope inclination was calculated with the GIS software based on the altitudes of the four corners of each mesh.

6.2.4 Statistical Analysis for Evaluating Habitat Suitability for Orangutans

Before developing the model to explain orangutan distributions, we calculated Spearman's rank correlations among nine of the ten factors, excluding D, which is binary, to detect autocorrelations. In this analysis, we divided the whole area of the reserve into 13,582 meshes of 180 m × 180 m and examined the correlations among the nine factors for the entire reserve. Significant internal correlations were found among the independent factors (Table 6.1), suggesting that single correlation analysis between the density of orangutan nests and the nine factors is not adequate to find independent factors that explain the variations of nest densities.

Table 6.1 Coefficients of Spearman's rank correlation among nine of the ten environmental factors, except for factor (D), history of reduced-impact logging

	(A)	(B)	(C)	(E)	(F)	(G)	(H)	(I)	(J)
(A) Distance from the main road	–	0.47*	0.53*	0.39*	0.23*	−0.60*	−0.07*	0.31*	−0.46*
(B) Distance from logging roads	0.47*	–	0.25*	0.20*	0.19*	−0.06*	−0.29*	0.00	0.18*
(C) Distance from all roads	0.53*	0.25*	–	0.39*	−0.04*	−0.44*	−0.07*	0.14*	−0.37*
(E) Distance from oil palm plantation	0.39*	0.20*	0.39*	–	−0.07*	−0.65*	0.07*	0.25*	−0.34*
(F) Distance from village	0.23*	0.19*	−0.04*	−0.07*	–	0.12*	−0.14*	−0.03	0.01
(G) Distance from natural lick	−0.60*	−0.06*	−0.44*	−0.65*	0.12*	–	−0.32*	−0.38*	0.49*
(H) Altitude	−0.07*	−0.29*	−0.07*	0.07*	−0.14*	−0.32*	–	0.44*	0.05*
(I) Slope inclination	0.31*	0.00	0.14*	0.25*	−0.03*	−0.38*	0.44*	–	−0.22*
(J) Distance from river	−0.46*	0.18*	−0.37*	−0.34*	0.01	0.49*	0.05*	−0.22*	–

*$P < 0.0001$

Considering the limitation of a linear model, we examined the effects of environmental factors on orangutan nests by a decision tree analysis (Breiman et al. 1984) with two options: one with the presence/absence of nests, and the other with density of nests. Three algorithms are generally used in decision tree analyses (i.e., CART, CHAID, and C5.0): we chose CART in our study. The CART algorithm uses a binary recursive partitioning approach to split a data set into nodes based on a single predictor variable chosen to minimize the deviation in the response variables in each of the resulting subsets. Nodes are added until they are homogeneous or contain too few observations. The variables that operate at large scales usually split the data early in the model, whereas variables that influence the response variable at more local scales operate later. When the response variable is factorial and binary, the proportion of the two factors in each terminal node is taken as the predictive value, in this case, the probability of the presence of orangutan nests. In our calculation, a minimum node size (number of observations) of 200 was used as the threshold value; nodes of this size or larger were adopted for a split.

6.3 Habitat Suitability for Orangutans

A total of 1,196 orangutan nests were observed in 673 meshes on the six census transects. Orangutan nests showed aggregative distributions around the plateaus in the southeastern side of the forest reserve (Fig. 6.2). Around the plateaus, nests tended to be found on the crest rather than on lower slopes.

The decision tree model demonstrated that two anthropogenic factors (main road and oil palm plantations) and two natural environmental factors (altitude and natural licks) significantly influenced the presence or absence of the orangutan nests (Fig. 6.3). The anthropogenic factors affected the presence or absence of the orangutan nests at larger scales than the natural environmental factors did because the anthropogenic factors became key factors at the early split in the model. This decision tree model resulted in five terminal nodes (P1–P5), two of which (P4 and P5) had high probability of the presence of orangutan nests. The meshes classified as P4 and P5 were far from the main road. The P4 meshes were close to natural licks (distance ≤7.1 km), and the P5 meshes were distributed at high altitudes (>118 m). Another two nodes, P1 and P2, had low probability of the presence of orangutan nests. The P1 meshes were close to the main road (≤7.2 km) and oil palm plantations (≤11.0 km). The P2 meshes were distributed at low altitude (≤118 m) and far from natural licks (>7.1 km), although they were far from the main road.

As for the density of orangutan nests, natural environmental factors (altitude, and distances from the natural licks and rivers) only were selected as the explanatory variables in the decision tree model for the analysis of 399 meshes in which orangutan nests occurred (Fig. 6.4). Meshes that did not include orangutan nests were excluded from the analysis. The decision tree model for the density of orangutan nests resulted in four terminal nodes (D1–D4), two of which (D3 and D4) had high nest densities. The D3 meshes were far from rivers (>435 m) and close to natural

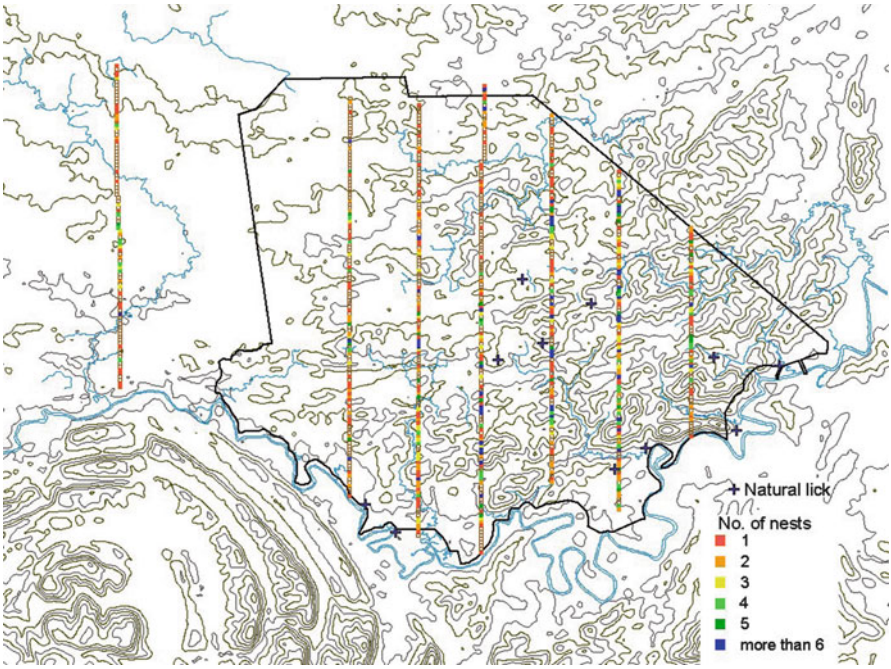


Fig. 6.2 Spatial distribution of orangutan nests drawn on a topographic map

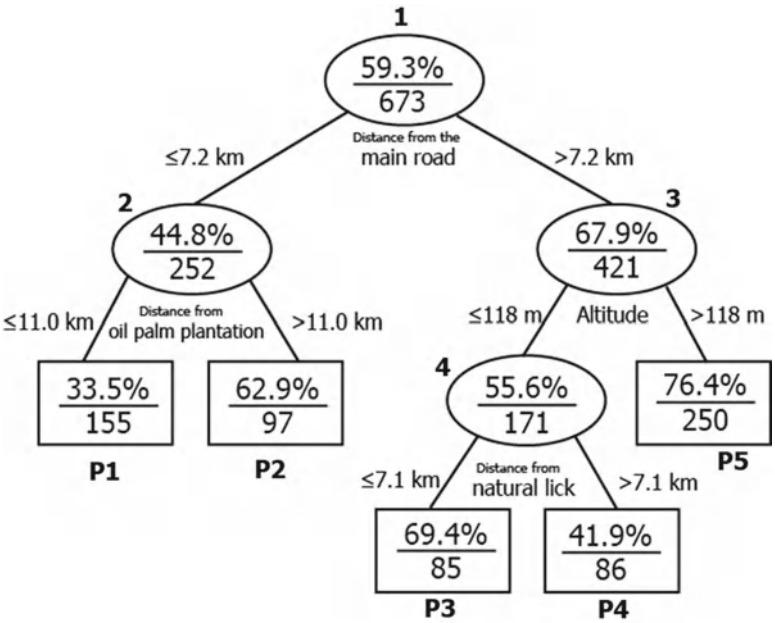


Fig. 6.3 Decision tree model for the presence/absence of orangutan nests. This model was valid for explaining the variation in nest distribution, because the AUC (area under the ROC curve) value was 0.79. The *numerator* at each node indicates the percentage of the number of meshes with orangutan nests relative to the total number of meshes (denominator) classified into the node. The *numerals beside the solid lines* indicate the threshold value of a key factor at each division. Terminal nodes are the resultant habitat types, namely, *P1–P5*

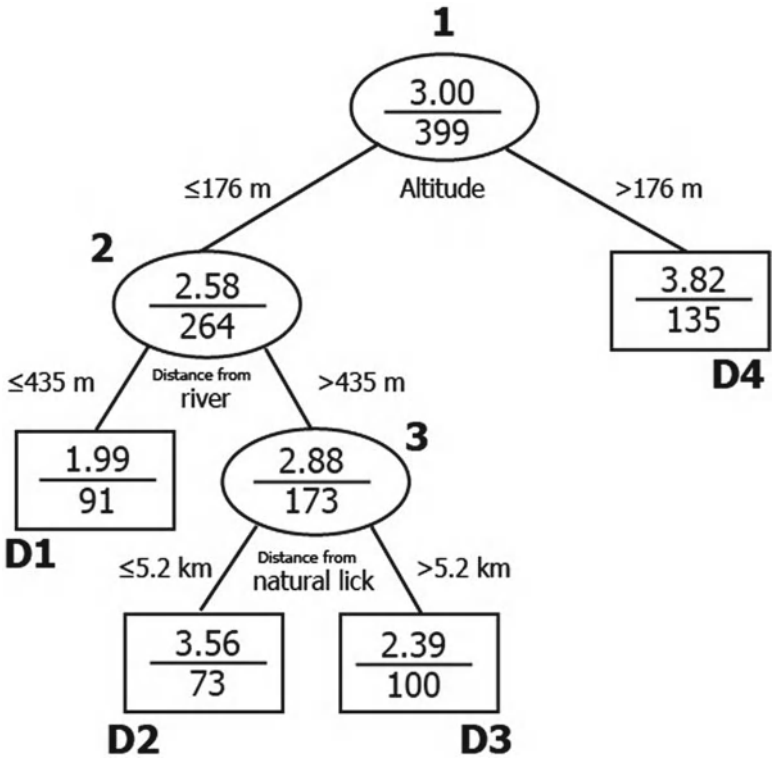


Fig. 6.4 Decision tree model for density of orangutan nests. This model was valid for explaining the spatial variation in the density of orangutan nests, because the AUC value was 0.76. The *numerator* at each node indicates the mean nest density (number of nests per 180 m × 180 m) for the meshes classified into the node (the number of meshes is expressed as the denominator). The *numerals beside the solid lines* indicate the threshold value of a key factor at each division. Terminal nodes are the resultant habitat types, namely, *D1–D4*

licks (≤5.2 km). The D4 meshes were distributed at high altitudes (>176 m). Another two nodes, D1 and D2, showed low nest densities. The D1 meshes were distributed at low altitude (≤176 m) and close to rivers (≤435 m). The D2 meshes were distributed at low altitude (≤176 m) and far from natural licks (>5.2 km). Distance from villages and logging history were not selected as significant factors in either model.

6.4 Guidelines for Establishing Conservation Areas

Artifacts and human activities, such as the main road with heavy traffic and oil palm plantations, had principal effects on the presence or absence of orangutans in the Deramakot Forest Reserve. Selection of these two factors (roads and plantations) in the model as a main decision factor indicates that habitat discontinuity affects the

distribution of orangutans. The discontinuity of forest canopies caused by the main road and oil palm plantations may prevent orangutans from migrating to adjacent areas and reduce their home ranges, because orangutans are arboreal mammals and avoid descending to the ground. Recent expansion of oil palm plantations can be a serious threat to orangutans in Borneo because it facilitates habitat fragmentation (Wich et al. 2008; Nantha and Tisdell 2009). Our results showed that effects of oil palm plantations on the presence of orangutans extended far, as much as 11 km inside the forest reserve (see Fig. 6.3). Based on this finding, we suggest that conservation areas inside the forest reserve be established away from the main road and oil palm plantations.

Some previous studies reported severe effects of logging on the presence of orangutans (Johns 1986; Felton et al. 2003, Morrogh-Bernard et al. 2003). In our analysis, the effects of logging could not be directly detected. Mode of logging operation in our study site (the Deramakot Forest Reserve) was reduced-impact logging. It is likely that this logging method did not cause enough perturbations to affect either the presence or the density of orangutans. Alternatively, the effects of reduced-impact logging could not be separated from the effects of the main road because reduced-impact logging had been carried out near the main road until the time of our study. Further studies are needed to fully understand the long-term effects of reduced-impact logging on the populations of orangutans.

Nest density was greater in the vicinity of natural licks. Because natural licks are important sources of minerals for mammals (Matsubayashi et al. 2007), natural licks may form hotspots of mammal diversity and abundance in the midst of tropical rain forests on highly weathered soils, where major cations are leached out. Orangutans are not an exception. They must rely on natural licks to intake minerals and consequently may form their home ranges to include natural licks. The area around a natural lick can be one of the most important areas for the conservation of orangutans. This need may be also true for other mammals, which should be substantiated in the future.

On the other hand, it was not clear why orangutans demonstrated greater nest densities in high-altitude areas as well as the areas far from rivers. It seems that they avoid low-lying areas along rivers, which are susceptible for flooding. We do not know the reasons exactly why they avoid such low-lying areas.

How could these results be used to establish conservation areas within production forests? In general, it is known that a single large area is superior to multiple small areas in conservation of large mammals, given the same total area (Diamond 1975). However, the whole area of the Deramakot Forest Reserve is subdivided a priori into small compartment units (approximately 500 ha on average), primarily for rotating timber production. Most compartments will be subjected to periodic logging in a 40-year cutting cycle in Deramakot. In such a setting, it is not readily possible to designate a single large conservation area inside the reserve. Alternatively, smaller multiple conservation areas may be suitable to Deramakot. In such a case, each of the multiple conservation areas must be highly suitable for orangutan

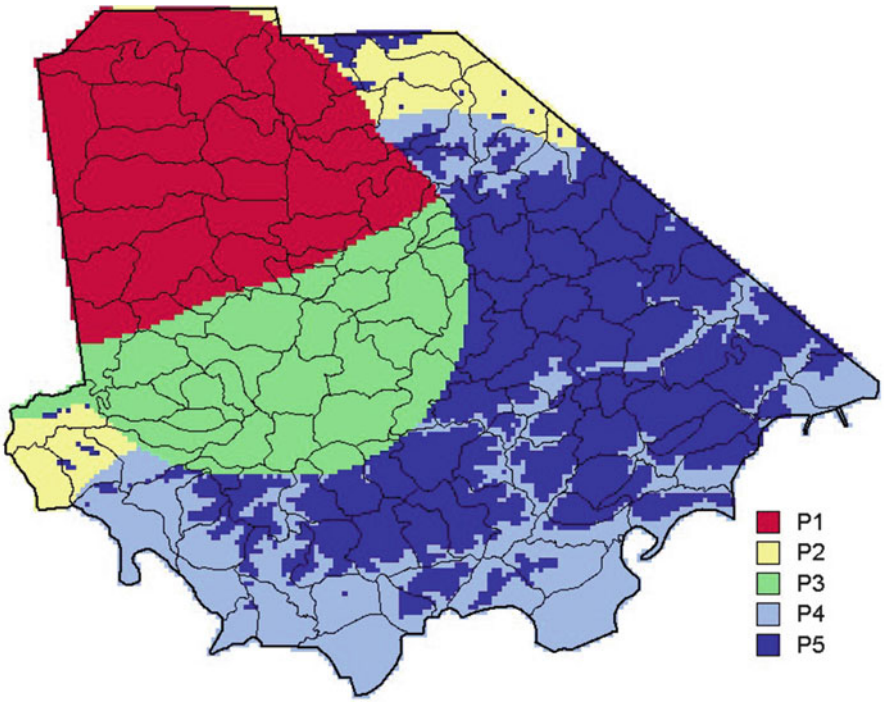


Fig. 6.5 Spatial distribution of habitat types classified by the decision tree model for the presence/absence of orangutan nests in Deramakot Forest Reserve. Legend for the habitat types is the same as in Fig. 6.3. The area designated as *P5* is most suitable for orangutan conservation, followed by *P4*. The area designated as *P1* is most unsuitable for orangutan conservation

habitats to enhance conservation efficiency. Moreover, such multiple conservation areas need to be dispersed over the entire forest reserve rather than being clustered to avoid the negative influences of a single logging event.

Our decision tree analysis can provide information to spatially delineate important areas to conserve populations of orangutans. The two decision tree models can indicate, on satellite data, combinations of spatial conditions that allow greater nest densities or nest presence. These areas are considered critical in the conservation of orangutans. Eventually, we can develop a map to show these critical areas. For example, we could draw two maps based on the results from the decision tree models, showing the distribution of habitat types, one with the probability of nest presence (Fig. 6.5) and the other with the estimated nest density (Fig. 6.6). In this procedure, we used Landsat ETM data as a base map. First, the Deramakot Forest Reserve was divided into 180×180 m meshes on the Landsat ETM data, and all explanatory variables that were selected in the models were estimated for each 180×180 m mesh. Subsequently, each mesh was categorized into a terminal node of the models.

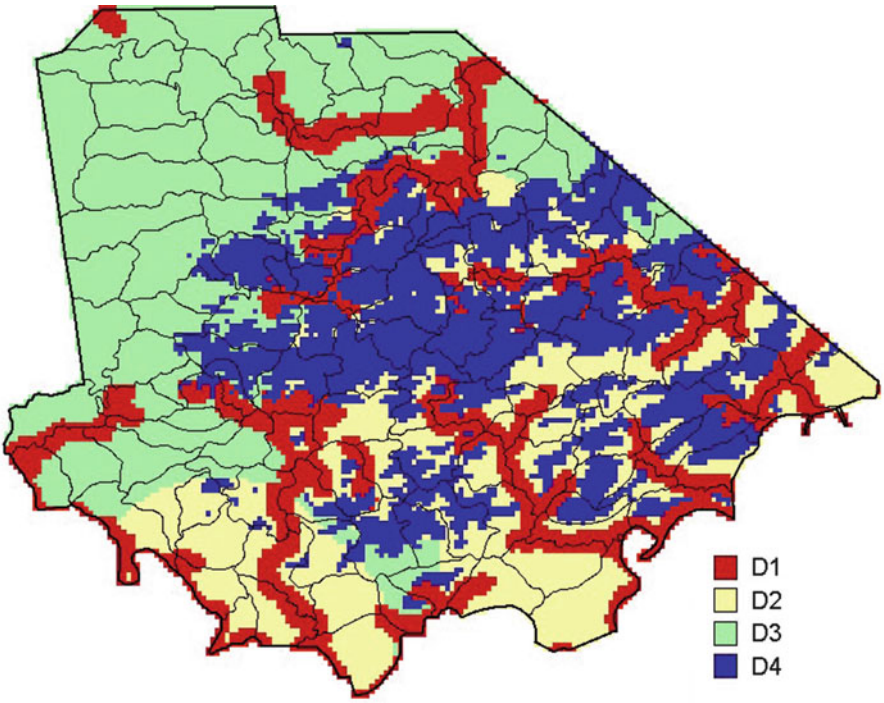


Fig. 6.6 Spatial distribution of habitat types classified by the decision tree model for the density of orangutan nests in Deramakot Forest Reserve. Legend for the habitat types is the same as in Fig. 6.4. The area designated as *D4* is most suitable for orangutan conservation, followed by *D3*. The area designated as *D1* is most unsuitable for orangutan conservation

The two maps in Figs. 6.5 and 6.6 demonstrate priority areas for the conservation of orangutans. Interestingly, the two maps demonstrate some discrepancies in priority area. For instance, the map based on presence/absence (Fig. 6.5) demonstrates that the northwestern side (indicated with red) of the Deramakot Forest Reserve is most unsuitable for the conservation area because the area is close to the main road and oil palm plantations. By contrast, the same area on the map based on nest density (Fig. 6.6) ranks second as suitable for the conservation area because the natural environments here are benign for orangutans (i.e., far from river and close to licks). The discrepancies suggest the inherent difference of the two models. The decision tree model based on presence/absence (Fig. 6.3) includes anthropogenic factors as an explanatory variable, suggesting that the model has incorporated unsafe sites to explain the absence of orangutans. By contrast, the decision tree model based on nest density (Fig. 6.4) includes natural factors only, suggesting that the model indicates natural habitat suitability. Considering this difference, we suggest selecting the areas most suitable for the habitat of orangutans first (based on Fig. 6.6) and then removing unsafe sites (based on Fig. 6.5) to designate conservation areas. Conservation areas for endangered mammals in production forests must satisfy both habitat suitability

and protection from anthropogenic factors, and can be designated by overlaying the two maps. Moreover, production forests are a dynamics system because of rotating logging operations, and our maps can be overlaid with harvest plans to avoid future negative influences of logging roads on orangutan populations.

Recently, many statistical analyses have been developed for the evaluation of habitat suitability (Hirzel et al. 2002; Engler et al. 2004; Thorstrom 2007; Trank et al. 2007). Among these analyses, the decision tree analysis, which we employed in our study, is useful for spatially delineating suitable habitats for the species of our concern. It is also useful to produce a set of policy suggestions because significant factors and their threshold values for habitat suitability (hence conservation priority) can be extracted.

We have demonstrated a numerical and quantitative procedure to map out suitable/safe habitats using orangutans as a model animal that can be incorporated into a long-term management plan of production forests to designate conservation areas. If mammals of critical conditions such as IUCN endangered categories inhabit within production forests, management bodies need to pay special attention to protecting these mammals. Our methods provide an example of a standard procedure to conserve such endangered mammals while maintaining timber production. In our analysis, logging history was not selected as a critical factor to determine the probability of orangutan occurrences in Deramakot, probably suggesting that current sustainable forest practices are not a major threat to orangutans. If logging is precautionarily planned to avoid most suitable habitats, as demonstrated in our maps (Figs. 6.5 and 6.6), conservation efficiency will further increase. However, this also needs to be discussed with reference to economic sustainability because the failure of sustainable forest management caused by economic loss will lead to overall failure in conserving threatened mammals. Our procedure will be advantageous in more highly degraded production forests with much reduced populations of endangered mammals than in Deramakot, because the protection of habitats is critical to maintain viable populations in such places. Our methods can be applied to other mammals, including ground-dwelling mammals, if presence/absence and abundance can be spatially elucidated. Automated sensor cameras (see Chap. 5), for instance, can be applied to our methods. Our methods should be widely adopted in many tropical production forests so that conservation areas can be established on the basis of appropriate data for understanding habitat suitability of focal species.

Topic Use of Natural Licks by Orangutans

By Hisashi Matsubayashi

The distance from natural licks was selected as an important factor in both decision tree models based on presence/absence or density of orangutan nests in Chap. 6. Why are orangutans attracted to natural licks? They may not come

(continued)

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down to natural licks unless there is a clear benefit for them because orangutans usually avoid descending to the ground. Natural licks of Deramakot occur sparsely throughout the reserve, especially at the base of cliffs and exposed ridges, and on road cuts. The parent materials of natural licks consist of sedimentary rocks. Most natural licks are small with an area of about 1 m^2, but some are larger, up to about 10 m^2 (Photo 6.1). At these licks, a small quantity of water seeps out from crevasses on the parental rocks and accumulates to a depth of about 1 cm. The water at these licks has a high concentration of minerals, such as sodium, potassium, calcium, and magnesium, as much as 10 to 100 times higher than nearby stream water. Natural licks are considered as an important source of minerals for orangutans and other mammals.

To investigate the frequency of mammal visitations to natural licks, a camera trap study was conducted from 2003 to 2009. Frequent visitors to the natural licks in Deramakot included some endangered large mammals such as banteng, Asian elephant, and orangutan (Matsubayashi et al. 2007). Matsubayashi et al. (2011) reported that several individuals of orangutan simultaneously shared the same natural licks. For example, four individuals were recorded in the same photograph, and all classes of maturity and sexes (flanged male, female with infant, and others, such as unflanged male or female without infant) were recorded on the same day. These results suggest that natural licks have a secondary function as a communication site in addition to a source of minerals.



Photo 6.1 An example of a natural lick in Deramakot

To confirm that the visitation of orangutans to natural licks is their general behavior, Matsubayashi (2009) studied natural licks in the Malua Forest Reserve. Malua is located at the opposite side of Deramakot, across the Kinabatangan River, the longest river in Sabah. Malua has a large population of orangutan (Ancrenaz et al. 2005), which is probably different from that of Deramakot because of the barrier formed by the large river. Frequent usage of natural licks by orangutans (Photo 6.2) was confirmed also in Malua. Visitation to natural licks is probably a common behavior among many orangutan populations in Sabah.



Photo 6.2 An orangutan drinking water at a natural lick in Deramakot, photographed with a sensor camera

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Chapter 7
Co-benefits of Sustainable Forest Management for Carbon Sequestration

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7.1 Introduction

Tropical forests sequester a large amount of carbon in the vegetation and soils (approximately 25% of the carbon in the terrestrial ecosystems) (Bonan 2008). The conversion of tropical forests through forestry and land-use change results in an enormous emission of carbon to the atmosphere. In the past decade, deforestation and forest degradation in Southeast Asia and elsewhere in the tropics accounted for approximately 20% of global anthropogenic carbon emissions (Gullison et al. 2007). The important roles of natural tropical forests in carbon storage and climate change mitigation are increasingly recognized.

One major cause of forest degradation in the tropics is unregulated selective logging, which usually damages more than 50% of the original forest biomass and causes surface-soil disturbances (Cannon et al. 1994; Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998). To reduce the deleterious logging impacts on the forests, reduced-impact logging has been developed and applied to a few tropical production forests (Kleine and Heuveldop 1993; Lagan et al. 2007; Putz et al. 2008a). Earlier studies have clearly and consistently demonstrated that reduced-impact logging can indeed reduce mechanical impacts on the forest stands and ecosystems (see also Chap. 3). As a co-benefit of reduced-impact logging, the forest retention of carbon can be

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improved by 30–50%, relative to unregulated (conventional) selective logging (reviewed by Putz et al. 2008a; Imai et al. 2009). This improvement implies that the implementation of reduced-impact logging is potentially useful to mitigate climate change through carbon emission reduction in addition to sustainable production of timber. A recent study by Putz et al. (2008b) suggests that carbon emission will be reduced by 0.16 gigatons carbon per year (approximately 10% of the total carbon emission from world tropical deforestation) if reduced-impact logging is implemented instead of conventional logging in the world's tropical forests designated for logging. However, these positive effects of reduced-impact logging on carbon pools are based primarily on the studies assessing the initial impacts of logging (e.g., comparison of above-ground biomass (AGB) between the pre- and post-logging periods) (e.g., Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998, 2003; Imai et al. 2009; but see Mazzei et al. 2010). Prolonged influences of reduced-impact logging need to be evaluated to develop a long-term forest management plan or ecosystem-based management approach. For instance, the estimate of the rate of recovery to a pre-logging biomass is the essential information for developing forest management plans. To date, only a few studies have demonstrated the long-term effects of reduced-impact logging on carbon pools in the ecosystem (Pinard and Cropper 2000).

Climate changes in terms of warming temperature (Lewis et al. 2009) and drought (Phillips et al. 2009) are predicted to affect carbon pools of tropical forests in the near future. Future forest management thus should take into account the effects of logging practices as well as climate change on forest growth.

In primary tropical rain forests on weathered soils, phosphorus (P) may constrain forest growth and hence the carbon (C) pool and its dynamics (Kitayama and Aiba 2002; Reich and Oleksyn 2004; Vitousek 2004; Wardle et al. 2004; Cleveland et al. 2006; Elser et al. 2007). Anthropogenic disturbances such as logging, which exports a large amount of P as timber from the ecosystem and as soil erosion, frequently strengthen P limitation (Lawrence et al. 2007). A recent study by Davidson et al. (2007) also reported that the loss of a large fraction of biomass nitrogen (N) provokes an N limitation and a conservative N cycle after disturbances. Therefore, the prediction of long-term forest growth and carbon dynamics should take into account the effects of the stoichiometric balance of ecosystem components (i.e., C/N, C/P, and N/P ratios of biomass, necromass, and soil), as well as the effects of climate change. Unfortunately, mathematical modeling is currently the only reliable way to predict long-term effects of the combination of nutrient limitation and climate change because empirical studies require an extended time period. Some individual-based growth models such as DIPSIM (Ong and Kleine 1995), FORMIX (Pinard and Cropper 2000; Glauner et al. 2003), FORMIX3 (Huth and Ditzer 2001), and SEIB-DGVM (Sato 2009), which consider growth, competition, regeneration, mortality, or gap formation, have examined long-term dynamics of tropical rain forests with varying logging scenarios. However, they cannot incorporate the effects of nutritional status.

The CENTURY model is a generalized ecosystem model that simulates C, N, S, and P dynamics for plant–soil systems and has been used extensively to simulate the impact of climatic and land-use changes on grassland, agricultural lands, forests, and savanna systems (Parton 1993; Schimel 1994; Parton et al. 1995; Schimel et al. 1996, 1997). In contrast to individual-based models, the CENTURY model can

simulate the combined effects of climate change and logging on carbon dynamics, taking into account the stoichiometric balance of ecosystem components.

The aim of this chapter is to examine the short- and long-term effects of reduced-impact logging on carbon sequestration in tropical rain forest ecosystems in Sabah, Borneo. First, we evaluated, in the field, the effects of two different logging practices (reduced-impact versus conventional logging) on the ecosystem carbon pools as the sum of above-ground vegetation, coarse and fine roots, necromass, and soil organic matter in Deramakot and Tangkulap. Second, we utilized the CENTURY model to examine long-term carbon dynamics. We collected information on climate, soil texture, plant productivity, and concentrations and amounts of C, N, and P in a remnant primary forest to parameterize the model. Then, we modified some input parameters within the model to approximate the condition of the primary forest studied. Finally, we utilized the CENTURY model to simulate carbon dynamics with several scenarios of varying harvest intensities, mimicking reduced-impact logging and conventional logging under climate change scenarios (warming air temperature and extreme drought) for the 100 years following a logging event.

7.2 Ecosystem Carbon Pools

7.2.1 Vegetation Survey

We established three or four 0.2-ha plots in each of the forests (primary, reduced-impact logged, and conventionally logged), as well as one 2-ha plot in each of the three forests in Deramakot and Tangkulap (see Chap. 3). All trees ≥10 cm diameter at breast height (dbh) were measured in each plot. AGB was estimated according to Brown's allometric equation (Brown 1997):

$$\mathrm{AGB} = \exp(-2.134 + 2.530 \ln\ \mathrm{dbh})$$

Coarse-root biomass (*Wr*) was estimated from an allometric relationship reported for a tropical lowland dipterocarp forest in Peninsula Malaysia (Niiyama et al. 2010):

$$Wr = 0.023\mathrm{dbh}^{2.59}$$

7.2.2 Sampling of Vegetation Components

To estimate the concentrations and amounts of C in above-ground vegetation, we collected samples of sun-leaves, bark, and wood from dominant canopy species. These ecological samples were collected from the dominant canopy species, defined as species with ≥3% relative basal area in the 2-ha primary forest and the 2-ha conventionally logged forest. Carbon concentrations of the vegetative components in the primary plot were used as substitutes for the values of the reduced-impact logged

plot because the species composition was similar between these two forests (see Chap. 3). Six *Shorea* species of Dipterocarpaceae (*S. atrinervosa*, *S. exelliptica*, *S. foxworthyi*, *S. macroptera*, *S. multiflora*, *S. parvifolia*) and one Anacardiaceae species (*Gluta wallichii*) were collected from the primary forest; four Dipterocarpaceae (*Dryobalanops beccarii*, *Parashorea malaanonan*, *Shorea macroptera*, *S. parvifolia*) and three *Macaranga* species of Euphorbiaceae (*M. conifera*, *M. gigantea*, *M. pearsonii*) were collected from the conventionally logged forest. The sum of relative basal area of these species attained 33.5% and 37.1% in the primary and the conventionally logged forests, respectively. Three trees were sampled from each of the species. Sunleaves were collected from the crown tops using a slingshot. From the same individual trees, bark and wood samples were extracted using a chisel and an increment borer at 1-m height. Wood cores were sampled from the outermost section within 3 cm from the bark. We collected at least three fully expanded mature leaves, and at least two samples of bark and wood per individual tree, and combined these samples by individual tree. In the laboratory, leaves were wiped, and all samples were oven-dried at 70°C for 3 days and then finely ground.

7.2.3 Soil Sampling

We excavated triplicate soil pits in each of three topographic positions [flat ridge (<10°), gentle slope (10–30°), and steep slope (>30°)] in each of the three 2-ha plots (i.e., nine soil pits per plot). Soil samples were collected from two O horizons (O1 and O2 horizon) and four mineral soil horizons (A, AB, B1, and Bt horizons). The O1 horizon represents the Oi horizon, comprising freshly fallen litter (twigs <2 cm girth, leaf and other fractions ≥~1 cm^2), and the O2 horizon represents Oe and Oa horizons, composed of more fragmented fractions with an accumulation of fine roots. The O horizons were sampled at three random points around each pit using a circular frame (23 cm diameter). At the same sampling point below the O horizons, A horizon (0–5 cm deep) was collected vertically with a 5-cm-deep core sampler (diameter, 37 mm). For each of the O1, O2, and A horizons, three samples per pit were combined to one composite sample (i.e., 3 composites × 3 topographic positions = 9 composites per horizon per plot). The other three horizons below 5 cm (AB, B1, and Bt horizons) were collected horizontally from the wall of the soil pit using the same core sampler. For each of AB, B1, and Bt horizons, one sample was collected per pit. Soils from three replicated pits per horizon were combined to one composite sample per topographic position (i.e., 1 composite × 3 topographic positions = 3 composites per horizon per plot). The boundary of A–AB was 5 cm deep, that of AB–B1 ranged from 16.3 ± 5.5 cm (primary) to 21.9 ± 5.7 cm (conventionally logged), and that of B1–Bt ranged from 41.1 ± 6.5 cm (primary) to 54.3 ± 6.0 cm (conventionally logged). The depth of the boundaries of AB–B1 and B1–Bt did not differ significantly among plots ($P > 0.05$ for both).

In nine of the eleven 0.2-ha plots, we set four 40-m line transects per plot, sampled O and A horizons by the same methods at 10-m intervals on each line transect, combined samples of each transect, and made four composite samples for each plot.

We sampled only O and A horizons in the 0.2-ha plots because carbon storage at deep horizons did not significantly differ among the three 2 ha-plots.

Samples of O horizons were sorted into soils, living roots, twigs, and leaves. The other mineral soil horizons were sorted into living roots and soils. All living roots <2 mm diameter were manually collected and gently rinsed with deionized water. Roots with dark-colored tissues or rotten bark were considered dead and were discarded. All composite soils were immediately stored in a refrigerator at 4°C until analysis. Twig and leaf litter and fine roots were oven dried at 70°C for 3 days, weighed, and finely ground.

7.2.4 Coarse Woody Debris

We estimated the mass of coarse woody debris (CWD) in the eleven 0.2-ha plots and an additionally established 0.2-ha subplot within the 2-ha primary plot. The diameter and length of all CWD ≥15 cm diameter fallen on the forest floor were measured in each plot. Standing dead trees ≥15 cm dbh were also measured for dbh and height. We classified all CWD according to its decomposition state into five classes (Harmon et al. 1995). The classes 1–5, from freshest to most rotten, were defined according to the following criteria. Class 1 material was newly fallen, solid wood with some leaves and/or fine twigs still attached. Material in class 2 was still solid and had intact bark but no fine twigs or leaves. Class 3 material resembled class 2 except that the bark of this class was rotten or sloughing. Class 4 material is somewhat rotten and could be broken when kicked. Class 5 material was rotten and friable, and it could be broken apart with bare hands.

Decayed wood samples were extracted using a power drill from the randomly selected CWD pieces. We collected at least four wood samples per decay class per forest (i.e., 4 samples × 5 classes × 3 forests = 60 samples). We could actually collect only four samples from class 1 because of its scarcity. For 11 pieces of class 5 debris that were too friable to sample with the power drill, we instead inserted a 5-cm-long core sampler (diameter, 37 mm) to a known depth and removed all material that filled the core sampler. The extracted samples ($n = 52$) were oven dried at 70°C, and wood density was calculated as oven-dry mass divided by fresh volume. The mass of CWD (Mg ha^{-1}) for each class was calculated from the CWD volume (m^3 ha^{-1}) and the specific density. Our estimation of the amount of C in the ecosystem does not include medium woody debris (≥2 cm girth and <15 cm diameter), trees <10 cm dbh, herbs, and lianas, which are a relatively small fraction in the entire ecosystem.

7.2.5 Chemical Analysis

We determined the concentration of the total C in the powdered samples of vegetation (sun-leaves, bark, and wood of canopy dominants) and below-ground components (standing leaf and twig litter, fine roots, and soil). We also determined the

concentrations of N and P of these components from the 2-ha primary plot for the parameterization of the CENTURY model. The C, N, and P concentrations of twig litter were used as substitutes for the values of CWD. The C, N, and P concentrations of wood were used as substitutes for the values of coarse roots. Soil subsamples were oven dried at 105°C for more than 48 h to determine gravimetric water content. The concentrations of total organic C and total N in samples were determined by the dry combustion method with an N-C analyzer (JM1000CN; J-Science Lab, Kyoto, Japan). To measure the concentrations of total P, subsamples of 0.2 g were weighed and digested on a block digester with concentrated H_2SO_4 and H_2O_2. Digestion was repeated until the solution became clear. Digests were filtered through Whatman 2 V filter paper and made up to 50 ml with deionized water. The concentration of P in the digests was determined on an inductively coupled plasma emission spectrometry (ICPS-7510; Shimadzu, Kyoto, Japan). The amounts of C, N, and P in each component on an area basis were calculated by multiplying the mass of each component by its corresponding weight-basis concentration.

Comparisons of mean (±SD) values of the C pools in the ecosystem components (AGB, coarse and fine roots, CWD, and soil) and of the C concentrations in soil horizons among forests were tested by analysis of variance (ANOVA). ANOVA was also used to compare the C, N, and P concentrations among vegetation components (leaves, bark, wood, and fine root) and among soil horizons (from O1 through Bt horizon). When the ANOVA P value was <0.05, the Tukey–Kramer post hoc test was performed to determine which pairs of means differed significantly.

7.2.6 Carbon Pools in the Current Forest Ecosystems

Mean concentration of C in vegetation components (leaves, bark, wood, and fine roots) varied within a narrow range from 41.3% to 50.6%, and was higher in above-ground components (leaves, bark, and wood) than in fine roots (Table 7.1). Soil organic C concentration drastically decreased with depth and ranged from 40.0% in O1 horizon to 0.2% in Bt horizon. In A and AB horizons, organic C concentration decreased with increasing logging intensity (primary ≥ the reduced-impact logged > the conventionally logged forest).

Carbon stock of AGB in the primary forest was estimated to be 236 ± 35 Mg C ha^{-1} (Fig. 7.1a). The conventionally logged forest stored 89 ± 38 Mg C ha^{-1} only as AGB, indicating that conventional logging reduced more than 62% of the initial above-ground carbon stock. By contrast, the reduced-impact logged forest stored 186 ± 31 Mg C ha^{-1} as AGB, corresponding to only 21% loss of the initial above-ground carbon stock. Coarse-root carbon stock also showed a similar pattern with AGB carbon (Fig. 7.1b). The amount of C in fine roots, CWD, and soils did not differ significantly among the three forests (Fig. 7.1c–e). The amount of C in the entire ecosystem (the sum of above-ground biomass, roots, necromass, and soils to 1-m depth) was 378, 318, and 195 Mg C ha^{-1} in the primary, the reduced-impact logged, and the conventionally logged forests, respectively (Fig. 7.1f). Conventional

Table 7.1 Concentration of carbon (C) in vegetation components and soil horizons in the three 2-ha plots

	C (%)								
	Primary		Reduced-impact		Conventional		N (%)		P (%)	
Vegetation components										
Leaf	50.6	±2.7			49.5	±2.0	1.90	±0.14	0.062	±0.012
Bark	48.6	±1.8			47.6	±1.7	0.57	±0.05	0.023	±0.003
Sapwood	49.1	±0.7			48.5	±0.4	0.40	±0.04	0.019	±0.002
Fine root	45.2	±4.4	41.3	±1.5	42.1	±1.0	1.38	±0.32	0.052	±0.004
Soil horizons										
O1	36.7	±1.5	40.0	±2.8	39.3	±2.5	1.14	±0.07	0.038	±0.005
O2	23.3	±9.6	22.3	±2.4	20.5	±4.0	1.00	±0.31	0.038	±0.003
A	3.6[a]	±0.5	2.7[ab]	±0.4	2.2[b]	±0.4	0.23	±0.02	0.027	±0.002
AB	1.4[a]	±0.2	1.3[ab]	±0.2	0.9[b]	±0.2	0.11	±0.001	0.026	±0.001
B1	0.6[a]	±0.08	0.4[b]	±0.03	0.7[a]	±0.14	0.07	±0.001	0.025	±<0.001
Bt	0.2[b]	±0.02	0.4[b]	±0.03	0.3[b]	±0.04	0.05	±0.002	0.025	±0.001
Other component										
Twigs in O horizons	42.1	±1.0	43.1	±1.9	39.9	±1.0	0.8	±0.15	0.032	±0.005

[a] Concentrations of nitrogen (N) and phosphorus (P) in the 2-ha primary plot are also shown

[b] Forests sharing the same letters do not differ significantly at $P<0.05$. Components without superscript (e.g., O1, O2, Twings) shows no significant difference ($P>0.05$) in C concentration between forest types by ANOVA.

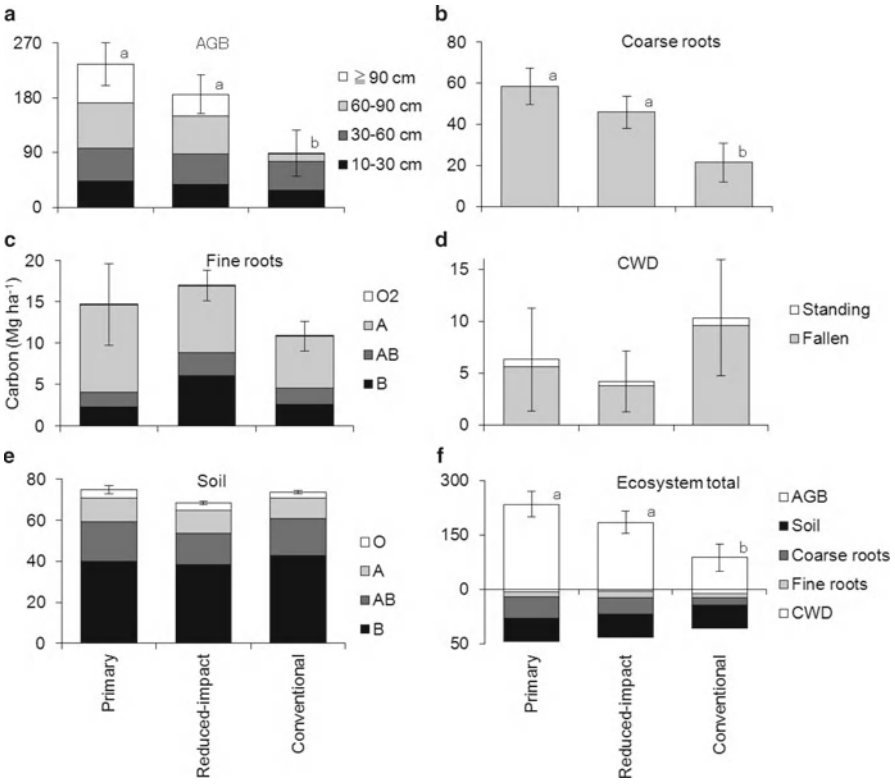


Fig. 7.1 Amounts of carbon in above-ground biomass (*AGB*) (**a**), coarse roots (**b**), fine roots (**c**), coarse woody debris (*CWD*) (**d**), soil organic matter (**e**), and the ecosystem total (**f**) in the three forests. Forests sharing the same letters do not differ significantly at $P<0.05$. *Error bars* indicate SD

logging reduced the initial ecosystem carbon pool by 49% at the time of our investigation (approximately 10 years after the logging), whereas reduced-impact logging reduced it by only 16%, indicating that reduced-impact logging could save carbon loss by 33% relative to conventional logging. It should be noted that this observation is based on our selected small plots.

7.3 Long-Term Carbon Dynamics

7.3.1 Model Description

We used CENTURY Soil Organic Matter Model version 4.0 in the forest mode (Metherell et al. 1993). The model is based on the cycles of C, N, P, and S each between vegetation and soils under a specific set of site conditions. The forest model has been tested for tropical, boreal, and temperate systems and used to simulate

forest system responses to different natural environmental variables, climate change, natural disturbances, and management practices (Sanford et al. 1991; Kelly et al. 1997; Raich et al. 2000; Yarie and Billings 2002).

To evaluate the long-term, combined effects of logging and climate change on the tropical rain forest of Deramakot, we first applied vegetation and soil data derived from our plots to parameterize the model. We then ran the model until the pools and flows of carbon in a tropical rain forest nearly stabilized, mimicking the first 2,000 years of ecosystem development using the climate parameters of Deramakot, and compared the outputs with observed values. We then modified parameters until simulated values converged to the observed values in the primary forest. We subsequently simulated the carbon dynamics for 100 years with four logging scenarios (as is shown in Sect. 3.3) combined with three climate change scenarios (current, warm, and warm + drought scenarios; see *Scenarios*).

7.3.2 Model Parameterization

Our work focused initially on the determination of parameter values that would generate quantitatively acceptable predictions of plant biomass, soil element pools, and ecosystem carbon pools. Input variables were calculated based principally on the data from our study (Table 7.1, Fig. 7.1) and Imai et al. (2010), and were supplemented by the literature data reported from the primary, lowland, or lower-montane tropical rain forests in Borneo (except for heath Kerangas forests). We modified parameters within the site.100 (environmental parameters) and tree.100 (physiological parameters) files, which were included in the CENTURY package.

At first, the variables in the Luquillo Experimental Forest, Puerto Rico (Sanford et al. 1991), which are the default values as the tropical forest within the model, were used as the initial input variables. Monthly average maximum and minimum air temperature and rainfall were obtained in Deramakot from March 2007 to July 2009 (Table 7.2), and these were set as the climate parameters within the site.100 file. Based primarily on the data from the 2-ha primary forest (Table 7.1, Fig. 7.1; Imai et al. 2010), C, N, and P pools in biomass by components (leaf, fine root, fine branch, large wood, coarse root, dead fine branch, dead large wood, and standing dead trees) were set within the site.100 file (Table 7.3). Soil C pools with different turnover rates (fast, intermediate, and slow), C/N and C/P ratios of soil organic matter, occluded P pool, soil pH, and bulk density were also set within the site.100 file (Table 7.3). The C/N and C/P ratios of biomass components, the death rate of forest components, leaf area control, and the element retranslocation rate were set within the tree.100 file (Table 7.4). Calculations of C pools by turnover rate, C/N and C/P ratios of soil organic matter, the death rate of forest components, and leaf area control are based on the formulas in the "CENTURY Parameterization Workbook." For the calculation of a leaf area control parameter (*btolai*), we optically measured the leaf area index (LAI) in the 2-ha primary forest using an LAI-2000 plant canopy analyzer (LI-COR; Lincoln, NE, USA). We placed five 100-m line transects 10 m

Table 7.2 Initial conditions of the climate parameters (*code names*) in the site.100 file of CENTURY model

Month	Temperatures (°C)						Precipitation (mm)		Precipitation (mm) recorded at 550 m in Mt. Kinabalu during 1998
	Minimum (*tmn2m*)		Maximum (*tmx2m*)		Mean		Mean (*precip*)	SD (*prcstd*)	
Jan.	22.0	± 0.5	29.7	± 0.7	24.4	± 0.2	273.0	± 127.9	148.8
Feb.	21.9	± 0.3	29.4	± 0.2	24.4	± 0.3	334.8	± 152.7	52.8
Mar.	22.0	± 0.3	31.5	± 0.9	25.2	± 0.7	179.8	± 116.1	28.5
Apr.	22.1	± 0.4	32.7	± 0.8	25.8	± 0.6	179.6	± 61.6	45.6
May	22.1	± 0.4	32.7	± 0.8	25.8	± 0.5	390.9	± 45.9	146.3
Jun.	21.9	± 0.5	32.3	± 1.0	25.5	± 0.6	261.8	± 72.5	227.7
Jul.	21.8	± 0.2	31.8	± 0.3	25.2	± 0.2	222.7	± 49.9	246.4
Aug.	21.4	± 0.2	32.0	± 1.1	25.2	± 0.5	216.8	± 179.1	200.3
Sep.	21.7	± 0.2	32.2	± <0.1	25.4	± 0.1	203.2	± 2.1	193.9
Oct.	22.0	± 0.1	32.3	± 0.3	25.5	± 0.1	249.5	± 82.9	119.2
Nov.	21.8	± 0.4	31.0	± 0.1	25.0	± 0.1	310.8	± 35.1	156.2
Dec.	22.4	± 0.4	30.3	± 0.2	24.8	± 0.3	275.1	± 19.3	239.5
Mean	21.9	± 0.3	31.5	± 1.1	25.2	± 0.5	258.1	± 64.1	–
Total	–		–		–		3097.7		1805.2

Precipitation of a severe El Niño drought year recorded at 550 m on Mt. Kinabalu during 1998 (Aiba and Kitayama 2002) is also shown

Table 7.3 Parameters (*code names*) used in the site.100 file of CENTURY model

Variables (*code names*)	Values	
Initial soil C pools (g C m^{-2})		
Surface OM with fast turnover (*somlci (1,1)*)		57
Soil OM with fast turnover (*somlci (2,1)*)		98
Soil OM with intermediate turnover (*som2ci (1)*)		3,018
Soil OM with slow turnover (*som3ci (1)*)		1,268
Plant residue at soil surface (*clittr (1,1)*)		223
Initial values for biomass C pools (g C m^{-2})		
Leaf (*rlvcis (1)*)		277
Fine root (*frtcis (1)*)		1,440
Fine branch (*fbrcis (1)*)		3,474
Large wood (*rlwcis (1)*)		18,458
Coarse root (*crtcis (1)*)		4,997
Dead fine branch (*wd1cis (1)*)		125
Dead large wood (*wd2cis (1)*)		266
Aboveground biomass (*aglcis (1)*)		22,208
Belowground biomass (*bglcis (1)*)		6,437
Standing dead trees (*stdcis (1)*)		587
Initial C/E ratios in soil OM	$i=1$, C/N	$i=2$, C/P
Surface OM with fast turnover (*rces1 (1,i)*)	16.1	483
Soil OM with fast turnover (*rces1 (2,i)*)	9.6	48
Soil OM with intermediate turnover (*rces2 (i)*)	20.1	112
Soil OM with slow turnover (*rces3 (i)*)	10.4	58
Surface litter (*rcelit (1,i)*)	96.6	2,897
Soil litter (*rcelit (2,i)*)	69.9	1,840
Initial values for biomass E pools (g E m^{-2})	$i=1$, N	$i=2$, P
Leaf (*rleave (i)*)	10	0.3
Fine root (*froote (i)*)	44	1.7
Fine branch (*fbrche (i)*)	28	1.4
Large wood (*rlwode (i)*)	150	7.2
Coarse root (*croote (i)*)	41	2.0
Aboveground biomass (*aglive (i)*)	189	8.9
Belowground biomass (*bglive (i)*)	84	3.6
Standing dead trees (*stdede (i)*)	12	0.4
Bulk density (g/cm^3) (*bulkd*)[a]		1.21
pH (H_2O) (*ph*)[a]		3.71
Initial value for occluded P (g P m^{-2}) (*occlud*)[a]		51.7

Initial conditions of pools of elements (E) (i.e., carbon (C), nitrogen (N), and phosphorus (P)), C/N and C/P ratios of soil organic matter (OM), occluded P pool, soil pH, and bulk density. The C pools, C/N, and C/P ratios of soil OM are deduced from the formulas in "CENTURY Parameterization Workbook", based on the data of Fig. 7.1 and Table 7.1

[a]Imai et al. 2010

apart from each other. We took ten LAI-2000 readings at 10-m intervals along each line transect and recorded an arithmetic mean of the ten readings. Mean (±SD) LAI was 8.09 ± 1.02, and this was used to calculate the *btolai* [biomass to LAI conversion factor (m^2 g^{-1})].

Table 7.4 Parameters (*code names*) used in the tree.100 file of CENTURY model

Variables (*code names*)	Values	
Initial C/E ratios of biomass components	*i* = 1, N	*i* = 2, P
Leaf (*cerfor (3,1,i)*)	27	816
Fine root (*cerfor (3,2,i)*)	33	870
Fine branch (*cerfor (3,3,i)*)	123	2584
Large wood (*cerfor (3,4,i)*)	123	2584
Monthly death rate fractions of forest components[a]		
Fine roots (*wooddr (2)*)		0.008
Fine branches (*wooddr (3)*)		0.004
Large wood (*wooddr (4)*)		0.001
Biomass to leaf area index (LAI) conversion factor ($m^2\ g^{-1}$) (*btolai*)[b]		0.015
N retranslocated from green forest leaves before litterfall (*forrtf (1)*)[c]		0.50
P retranslocated from green forest leaves before litterfall (*forrtf (2)*)[c]		0.83

Initial carbon (C)/element (E) ratios of biomass components, the death rate of forest components, leaf area control, and the element retranslocation rate are shown

The death rate of forest components and leaf area control are deduced from the formulas in "CENTURY Parameterization Workbook"

[a]Based on the above-ground biomass increment rate (Seino et al., (2005)), and the data from Fig. 7.1, Table 7.1; Proctor et al. (1983), Pendry and Proctor (1996), Burghouts et al. (1998), Silver et al. (2005), Hertel et al. (2009), and Leuschner et al. (2009)

[b]The estimated mean LAI (±SD) in the 2-ha primary forest was 8.09 ± 0.12, and the data of leaf biomass were from Imai et al. (2010)

[c]Based on Table 7.1, Proctor et al. (1983), Pendry and Proctor (1996), Burghouts et al. (1998), Kitayama and Aiba (2002), and Kitayama et al. (2000, 2004)

Allocation patterns (*fcfrac*) and the death rates of forest components (leaf, *leafdr*; wood, *wooddr*) strongly regulate forest biomass and productivity, but it is difficult to measure these values in the field. We used the default values of *fcfrac* and *leafdr* because of the deficiency of data collected. We adjusted the *wooddr*, which can be estimated based on our data and literature data, until the C pools in forest live components matched observed values. Monthly death rate (*wooddr (i)*) of fine root (*i* = *2*), fine branch (*i* = *3*), and large wood (*i* = *4*) were set as 0.008, 0.004, and 0.001, respectively (Table 7.4). These values are consistent with the rates calculated from our data and the literature data; the estimated *wooddr 2*, *3*, and *4* range was 0.004–0.007, 0.002–0.004, and 0.001–0.002, respectively.

Given these tunings, we were able to determine appropriate parameters for C pools after a 2,000-year period of ecosystem development mimicking the primary forest under the current climate of Deramakot. Simulated total C in forest live components (*frstc*) was 286 Mg C ha^{-1}, which exactly matched the observed value of 286 Mg C ha^{-1}; total C in the forest ecosystem (*fsysc*) was simulated to be 340 Mg C ha^{-1} in comparison with the observed 334 Mg C ha^{-1}; soil organic carbon in the top 20 cm (*somtc*) was simulated to be 31 Mg C ha^{-1} in comparison with the observed 41 Mg C ha^{-1}; necromass (CWD and dead roots) carbon (*wood1c* + *wood2c* + *wood3c* + *stdedc*) was simulated to be 22 Mg C ha^{-1} in comparison with the observed 6–9 Mg C ha^{-1} (C pool in CWD was observed value; the dead root C was actually estimated using the "CENTURY Parameterization Workbook").

Table 7.5 Drought scenario and four logging scenarios used in the model

		Reduced-impact		Conventional	
	Drought	5%	20%	50%	60%
Fractions removed from pools					
Live leaves (*remf (1)*)	0.015	0.05	0.2	0.5	0.6
Live fine branches (*remf (2)*)	0.015	0.05	0.2	0.5	0.6
Live large wood (*remf (3)*)	0.015	0.05	0.2	0.5	0.6
Dead fine branches (*remf (4)*)	0	0	0	0	0
Dead large wood (*remf (5)*)	0	0	0	0	0
Fractions of live root components that die					
Fine root (*fd (1)*)	0.015	0.05	0.2	0.5	0.6
Coarse root (*fd (2)*)	0.015	0.05	0.2	0.5	0.6
Fraction of E in killed forest components that is returned to the system					
Leaves (*retf (1,i)*)	1	1	1	1	1
Fine branches (*retf (2,i)*)	1	1	1	1	1
Large wood (*retf (3,i)*)	1	0	0.75	0.7	0.667

7.3.3 Scenarios

We made three climate and four logging scenarios to simulate the combined effects of logging and climate changes on the carbon dynamics during 2000–2100. IPCC (2007) predicted that air temperature in Southeast Asia will increase by 2.5°C from the 1980–1999 period to the 2080–2099 period under the A1B scenario. The A1B emission scenario is balanced across all energy sources such as fossil and non-fossil energy sources, and many of the A1B results are assessed in the IPCC 2007 report. Therefore, the scenario, which incorporates the increases of the average minimum and maximum air temperature by 0.025°C per year during 2000–2100, was selected as the "warm" scenario. We also assumed that the climate change would be associated with extreme El Niño droughts. From 1997 to 1998, Southeast Asia was affected by a severe drought associated with the El Niño Southern Oscillation (ENSO) that was the strongest on record in terms of sea-surface temperature anomalies (McPhaden 1999). Aiba and Kitayama (2002) reported that tree mortality increased from 1.1% at a pre-drought period (1995–1997) to 2.6% at the drought period (1997–1999) at 550 m on Mt. Kinabalu. We thus set the "warm + drought" scenario, which consists of the "warm" scenario and the regular occurrence of droughts at 10-year intervals. In each drought year, total monthly precipitation recorded at 550 m on Mt. Kinabalu during 1998 was used as a precipitation parameter (*precip*) (Table 7.2). At the same time, above-ground C was adjusted to decrease by 1.5% mimicking the increased mortality caused by the drought. In the latter, we assumed that 100% of dead vegetation components would return to the ground (Table 7.5). The "current" scenario assumes that the mean air temperature and precipitation in 2000 will persist until 2100.

Four logging scenarios with different logging intensities were made (Table 7.5), based on our results and literature data (Pinard and Putz 1996; Bertault and Sist

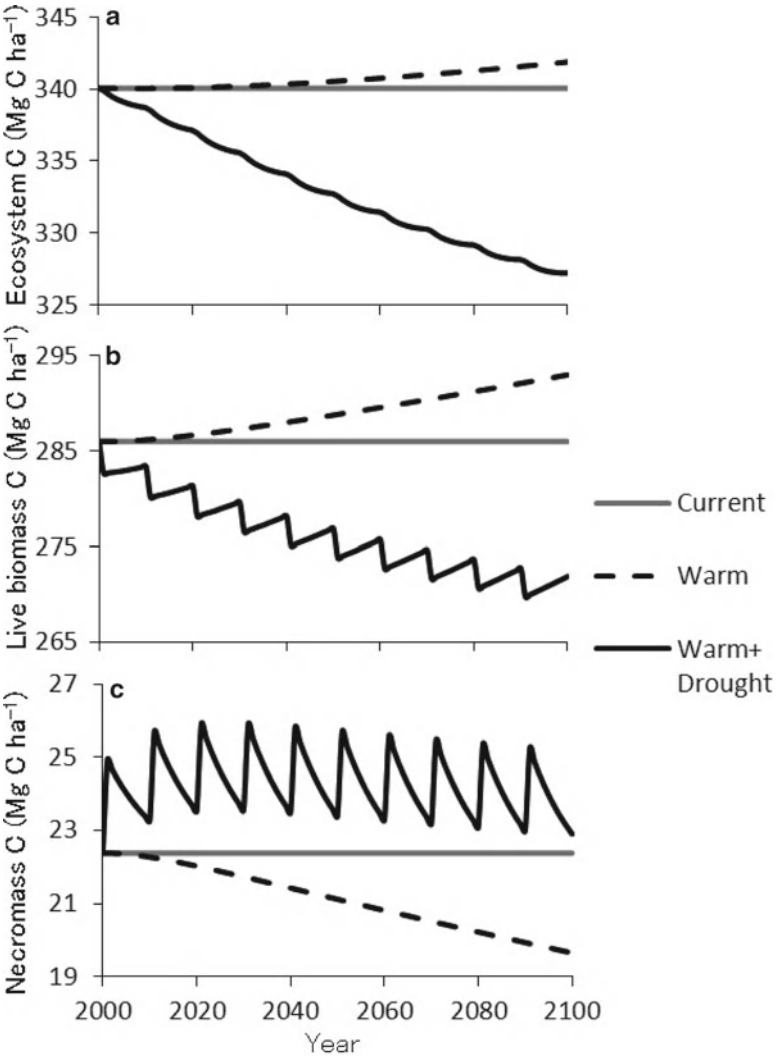


Fig. 7.2 Simulated changes of carbon (C) pools of the entire ecosystem (**a**), live biomass (**b**), and necromass (**c**) for the three climate scenarios during 2000–2100. The current scenario, persisting current climate condition; the warm scenario, increasing monthly average maximum and minimum air temperature at 0.025°C year^{-1}; the warm + drought scenario, droughts equivalent to the 1997–1998 El-Niño at 10-year intervals in addition to the "warm" scenario

1997; Sist et al. 1998), which compared AGB between reduced-impact logged and conventionally logged forests. Reduced-impact logging usually reduced the original AGB by 20–25%, whereas conventional logging reduced the original AGB by 50–60%. Such a decline of AGB includes the amount of both exported timber and collaterally killed trees during road construction and logging. Sabah Forestry Department (2005) has planned the average timber yield by reduced-impact logging

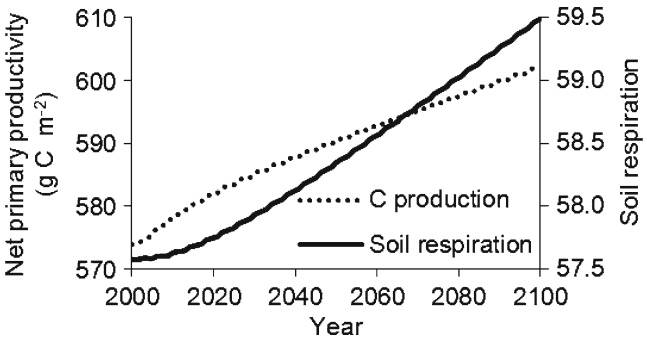


Fig. 7.3 Simulated changes of net primary productivity (carbon production, *cproda*) and soil respiration (*amt1c2*) under the "warm" scenario during 2000–2100

not to exceed 40 m^3 ha^{-1}. On the other hand, conventional logging may export 100–170 m^3 ha^{-1} of timber from the ecosystem, according to previous studies (Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998). Assuming that the average wood density is 0.6 $g\,cm^{-3}$ (Slik et al. 2008), the exported amount of AGB by logging represents approximately 5% of the original stand in reduced-impact logging, whereas it is 13–23% in conventional logging. Therefore, the percentage AGB reduction from the original stand was set as 5%, 20%, 50%, and 60%, and the percentage AGB exported from the ecosystem was set as 5%, 5%, 15%, and 20%, respectively; the remaining 0%, 15%, 35%, and 40% of AGB, respectively, was assumed to return to the ecosystem. These four sets of percentages correspond to the reduced-impact logging without any collateral damage (5% reduced-impact), the common reduced-impact logging (20% reduced-impact), the common conventional logging (50% conventional), and the highly destructive conventional logging (60% conventional), respectively (Table 7.5). We incorporated these logging scenarios in a trem.100 file in the model, assuming that all logging events occurred in the primary forest only once in year 2000, and compared the subsequent carbon dynamics until year 2100 among the scenarios.

7.3.4 Long-Term Effects of Logging on Carbon Pools

When the "warm" scenario was applied, the amount of live biomass C and entire ecosystem C gradually increased, whereas the amount of necromass C gradually decreased from 2000 to 2100 (Fig. 7.2) because the increased temperature promoted the net primary productivity and decomposition rate (Fig. 7.3) (Kitayama and Aiba 2002). However, when the "warm + drought" scenario was applied, the amounts of ecosystem C and biomass C decreased while the amount of necromass C increased (Fig. 7.2).

Figure 7.4 shows the C dynamics with the four logging scenarios under the "warm + drought" scenario. Conventional logging (CV 50% and 60%) remarkably decreased

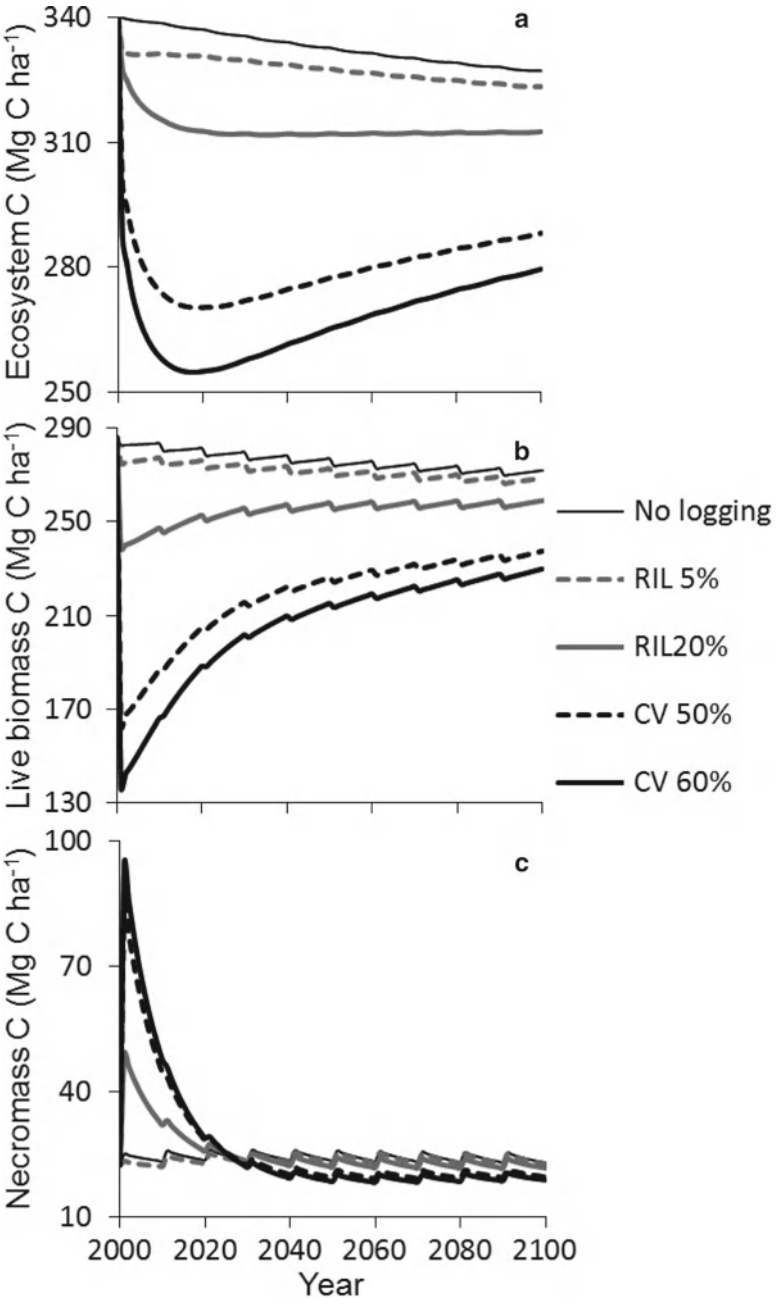


Fig. 7.4 Simulated changes of carbon (C) pools in the entire ecosystem (**a**), live biomass (**b**), and necromass (**c**) for the four logging scenarios (5%, 20%, 50%, and 60% of forest biomass removal) under the "warm + drought" climate scenario during 2000–2100. RIL 5%, reduced-impact logging with 5% AGB reduction and 5% export; RIL 20%, reduced-impact logging with 20% AGB reduction and 5% export; CV50%, conventional logging with 50% AGB reduction and 15% export; CV60%, conventional logging with 60% AGB reduction and 20% export. Refer to Table 7.5 for details of the scenarios

the live biomass C (Fig. 7.4b) and produced a flush of necromass (Fig. 7.4c). The necromass gradually decreased (i.e., decomposed) and returned to a basic level of the primary forest in 20–30 years (no logging) (Fig. 7.4c). Thus, the net carbon emission from the ecosystem persisted during the first 20–30 years (Fig. 7.4a). If conventional logging takes place, a longer time period than 100 years will be required for biomass and ecosystem C pools to recover to the original state. On the other hand, reduced-impact logging (RIL 5 and 20%) did not largely change biomass, ecosystem, and necromass C compared with conventional logging. Carbon pools in the reduced-impact logged forest will therefore recover more quickly to the original state.

7.4 Concluding Remarks

Our model predictions demonstrated the combined effects of logging practices and climate changes on the long-term carbon dynamics for a first approximation. Air temperature increase at the rate of 2.5°C during the next 100 years in Southeast Asia (IPCC 2007) gradually increased net primary productivity, forest biomass C, and ecosystem total C from 2000 to 2100. This result is consistent with the recent evidence that temperature warming increases carbon pools in African (Lewis et al. 2009) and Amazonian tropical forests (Baker et al. 2004; Phillips et al. 2008). The combination of the air temperature warming and the regular occurrence of extreme droughts gradually decreased the ecosystem C pools during 100 years. Extreme droughts are known to decrease forest biomass in tropical rain forests through increased tree mortality (Aiba and Kitayama 2002; Phillips et al. 2009). When conventional logging was applied to the forest under warming and droughts at regular intervals, the biomass and ecosystem C pools dramatically decreased, and the effects persisted for more than 100 years. On the other hand, reduced-impact logging did not drastically decrease these C pools under the same climate scenarios. These findings indicated that logging intensity is the most important factor affecting the long-term C dynamics, rather than climate changes. The initial logging dictates the long-term carbon dynamics, indicating that "reduced-impact" logging with reduce timber removal can maintain a greater amount of carbon throughout the long-term recovery phase.

After a 40-year period, which is the typical logging cycle in Sabah, carbon pools in the forest harvested with reduced-impact logging are predicted to be at least 37 Mg C ha^{-1} greater than those in the conventionally logged forest (comparison of the ecosystem C pools between RIL20% and CV50% in 2040). Our prediction may be conservative and the actual difference can be much higher. Putz et al. (2008b) reported that the benefit of reduced-impact logging over conventional logging was 30 Mg C ha^{-1}, and this value is similar to our prediction. Based on this value, Putz et al. (2008b) further suggested that reduced-impact logging applied to all tropical forests designated for logging would reduce carbon emissions by about 10% compared with destructive logging operations. We also suggest that a substantial reduction of carbon emission can be expected if reduced-impact logging is applied widely to tropical production forests while timber is sustainably produced.

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Chapter 8
Synthesis: Co-benefits of Sustainable Production Forestry

Kanehiro Kitayama, Robert C. Ong, and Ying Fah Lee

8.1 Introduction

Comparisons between the consequences of improved management and those of conventional logging in terms of carbon stock and biodiversity conservation were conducted in the Deramakot Forest Reserve versus the Tangkulap Forest Reserve, where lowland dipterocarp forest is the natural vegetation (see also Imai et al. 2009). In this chapter, we summarize the co-benefits of improved forest management for biodiversity conservation and carbon sequestration. We evaluate the co-benefits in terms of additionality, which is the net increment effect of the improved forest management added over 13 years from 1989 (when logging was suspended in Deramakot) until 2002 (when our analyses started) in comparison to continued high-impact conventional logging.

Briefly, Deramakot and the surrounding areas were licensed for logging from 1955, and "conventional logging" with a heavy impact was applied in the area (see Chap. 1 for details). The Sabah Forestry Department, with technical support from the German Aid Agency for Technical Cooperation, began developing a forest management plan in Deramakot in 1991. All logging activities were suspended in Deramakot in 1989, but continued in Tangkulap, an adjacent management unit. With the implementation of sustainable forest management practices in Deramakot beginning in 1995, reduced-impact logging (RIL) was adopted in harvesting operations as a standard practice, with minimal impact on the physical environments

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since then (Kleine and Heuveldop 1993; Lagan et al. 2007). However, Tangkulap was continuously harvested under more destructive, high-impact conventional logging methods until 2001. Although sustainable forest management was initiated in 1995, for estimation of the net carbon and biodiversity gain, regrowth of the forest after 1989 needs to be taken into account. In this case, we can consider the conventional logging practice in Tangkulap as the baseline scenario (business as usual), a practice that has been typically applied widely throughout Borneo.

8.2 Synergy of Carbon Sequestration and Biodiversity Conservation in Improved Sustainable Forest Management

8.2.1 Carbon sequestration

Mean carbon density in Deramakot and Tangkulap was evaluated by three different methods in this volume:

1. Snapshot analysis in the same year using satellite remote sensing at a landscape level (Chap. 2)
2. Time-series analysis using satellite remote sensing at a landscape level (Chap. 2)
3. Snapshot analysis using field sample plots in forest stands subjected to different logging practices (Chap. 7)

In the first method (see Chap. 2), a newly developed algorithm was applied to Landsat ETM data taken in 2000, and the above-ground biomass was estimated for each of all 30 × 30 m pixels, except for those that were covered with cloud and haze. The amount of carbon was further calculated by assuming that 49% of the above-ground biomass was carbon. The mean carbon density (Mg/ha) in above-ground biomass in the year 2000 was 168 (range, 144–196) in Deramakot and 123 (range, 105–143) in Tangkulap. Deramakot had a greater carbon density by 45 Mg/ha than Tangkulap as of 2000.

In the second method (see Chap. 2), the same algorithms was applied to Landsat TM or Landsat ETM data taken in 1991, 2000, and 2007. Net change in carbon density was investigated in Deramakot between 2000 and 2007 (corresponding to the period of improved management, i.e., restricted harvest in two to three compartments with RIL) and in Tangkulap between 1991 and 2000 (corresponding to the period of conventional logging throughout the area). A time-series analysis was applied to two different periods because time could not be controlled for management effects. Tangkulap lost 6 (5–7) Mg C/ha per year between 1991 and 2000 when conventional logging was the mode of harvest. By contrast, Deramakot gained 6 (5–7) Mg C/ha per year between 2000 and 2007 when RIL was the mode of harvest. This finding translates to a net increment of 12 Mg C/ha per year as the additionality of improved management over the baseline logging practice.

In the third method (see Chap. 7), one 2-ha and three or four 0.2-ha research plots were subjectively placed in each of the forest that was harvested with RIL versus the forest that was harvested by conventional logging methods. Data were collected in 2006–2008 (mean, 2007), which corresponded to 7–10 years after the last RIL in the former and at least 7 years after the last conventional logging in the latter. The primary forest without evidence of any past logging activities was also investigated as a reference. The above-ground biomass within each plot was quantified by making an inventory of trees with a diameter at breast height (dbh) of greater than 10 cm, using an allometric equation. Assuming that 49% of biomass is carbon, the mean ± SD carbon density (Mg/ha) of the above-ground vegetation was 236 ± 35 for the primary forest, 186 ± 31 for the reduced-impact logged forest, and 89 ± 38 for the conventionally logged forest. Mean carbon density was significantly greater in the reduced-impact logged forest than in the conventionally logged forest by 97 Mg/ha. Mean carbon density was not significantly different between the primary versus the reduced-impact logged forests.

We are dealing with a highly dynamic system where regrowth and harvest are simultaneously occurring. The entire 55,149-ha Deramakot area is divided into 135 compartments, and only two or three compartments are harvested each year. Under the current long-term forest management plan, each compartment will theoretically receive RIL once in 40 years and will be allowed regrowth for 39 years, because the mean cutting cycle is assumed to be 40 years. As the second method of a time-series analysis indicated, the regrowth of the fallow compartments will surpass the harvest taking place in a restricted number of compartments by 6 Mg C/ha per year in Deramakot.

The net carbon increment in sustainable forest management and the net carbon loss in conventional logging in a landscape context can explain the snapshot difference of 45 Mg/ha for the year 2000 between Deramakot and Tangkulap in the remote sensing analysis. On the other hand, the plot analysis depicted a greater difference (97 Mg/ha) of mean carbon density in a forest stand context. The difference of 97 Mg/ha probably reflects the effects of localized harvest intensity and collateral damage to standing biomass. Moreover, the effects of regrowth in fallow compartments are not taken into account in the plot analysis.

Insofar as current sustainable forest management is practiced, Deramakot will continue to sequester carbon at a rate of about 6 Mg C/ha per year for a long time until the above-ground biomass becomes saturated at a value equivalent to the reference primary forest (i.e., 236 ± 35 Mg/ha). This result is consistent with our ecosystem model simulation (Fig. 7.4; Chap. 7). By contrast, forest stands subjected to high-impact conventional logging will require an extremely long process of recovery and will not likely approach the value of the reference primary forest, even in 100 years.

All analyses consistently suggest a beneficial effect of improved forest management for the retention and sequestration of carbon. If we consider the snapshot difference of 45 Mg/ha for the year 2000 as a legitimate value, improved management resulted in a net addition of 2.5×10^6 Mg carbon for the 55,149-ha area of Deramakot for 11 years after 1989 when all logging was banned and sustainable forestry commenced.

8.2.2 Biodiversity Conservation

In our study area, community composition and taxonomic richness were analyzed for trees, flying insects, medium-to-large mammals, soil macrofauna, wood-decaying fungi, and soil microbes. These taxonomic groups were compared among the primary forest, reduced-impact logged forest, and conventionally logged forest to elucidate the effects of management.

The richness of species, genera, and families of canopy trees per small plot (0.2 ha) and several indices of species diversity were consistently and significantly greater in the reduced-impact logged forest than in the conventionally logged forest, but were not different between the reduced-impact logged forest and the pristine forest (see Chap. 3). The maintenance of the richness of tree species in the reduced-impact logged forest may be attributed to the recruitment of "newcomers" that invaded patches mildly disturbed by RIL. However, the community composition of the reduced-impact logged forest was similar to that of the nearby primary forest. The conventionally logged forest differed from the reduced-impact logged forest and the primary forest in community composition. Therefore, it is safe to conclude that sustainable forest management with RIL can maintain the integrity of community composition, but that conventional logging has a negative influence.

Another example is soil macrofauna, which is representative of decomposers exceeding 2 mm in body size. The density and the number of taxa of soil macrofauna at the order or an equivalent taxonomic level were not different among the reduced-impact logged, conventionally logged, and primary forests (see Chap. 4), which is in contrast to the tree species community. However, the composition of the soil macrofauna community was modified greatly by conventional logging but less so by RIL. Consequently, sustainable forest management with RIL could maintain the integrity of the soil macrofauna community composition, as similarly found with the tree species community. The same pattern was confirmed also for wood-decaying fungi, flies relying on wood debris, ants, and soil microbes (see Chap. 4). Sustainable forest management could maintain a greater community integrity also for flying insects (Akutsu et al. 2007).

A survey using camera traps (see Chap. 5) showed that the abundance of seven mammal or bird species was greater in the sustainably managed forest than in the conventionally logged forest. Medium to large mammals are often hunted for bush meat unless the access of hunters is physically limited. The greater mammal abundance in the sustainably managed forest may reflect the improved physical condition of the forests as well as the level of protection because of restricted access to hunters within these forests. Another independent helicopter census of the orangutan population also indicated a significantly higher nest density in the sustainably managed forests than in the surrounding conventionally logged forests (Ancrenaz et al. 2005). Interestingly, logging history within the sustainably managed forest of Deramakot was not selected as a variable in the decision tree models to explain the presence/absence or density of orangutans in Chap. 6. Therefore, the implementation of logging per se does not cause deleterious impacts on orangutan populations if improved logging practices are applied, combined with a sufficient number of fallow

compartments, which may become a refugium for them; this may be true also for other medium to large mammals, as demonstrated in Chap. 5.

These results consistently indicate that improved forest management in Deramakot has beneficial effects on biodiversity. Strict compliance with good management principles maintains the structural diversity of vegetation in each stand, the abundance of keystone fruit species, standing dead snags, large stems, foliage, and litter on which animals depend for food and habitat (Johns 1997), and probably, these all together help in maintaining the communities of plants and animals relatively intact. It should also be emphasized that the number of the compartments harvested each year is restricted and hence logging impacts are localized in Deramakot. The remaining compartments are allowed for regrowth and become a refugium for, or a source of, biological species. The sustainable forest management in Deramakot has two types of beneficial effects on biodiversity: (1) locally reduced impacts in each of the forest stands subjected to RIL and (2) refugium/source effects in the vast fallow compartments. How each of the two effects has contributed to the overall conservation of biodiversity cannot be determined in our analyses. It seems that less-mobile biological species (plants, insects, fungi, soil fauna, and soil microbes) benefit from locally reduced logging impacts (Chaps. 3 and 4), whereas more-mobile species (mammals and birds) are benefited by both effects (Chaps. 5 and 6). This research question must be substantiated in the future to develop better management options.

Considering that both Deramakot and Tangkulap were the target of logging activities between 1955 and 1989, the plant and animal communities recovered their integrity in Deramakot after the suspension of conventional logging. Biodiversity has therefore responded to the avoidance of forest degradation in a reasonably short time period. This observation is not meant to suggest that biological assemblages can recover in any production forests irrespective of the magnitude of forest degradation. We indeed do not know how quickly the biological assemblages in present-day Tangkulap can recover.

8.3 Technical Issues in Monitoring Biodiversity

Logging can directly influence tree species diversity. Surprisingly, heavy logging did not reduce the number of tree species per unit number of stems (see the species accumulation curve in Fig. 3.3k). The loss of commercial species from harvesting was most likely complemented by the addition of fast-growing pioneer species (Fig. 8.1). If the species number of such recruits after logging surpasses the number of eliminated species in a logged-over forest, the total number of species can increase as compared with the initial condition (Cannon et al. 1998). Therefore, the number of tree species (or genera or families) per unit area/stem should be interpreted with precaution, notwithstanding that the species number is the most widely used first approximation of species diversity. A similar misleading effect can occur within any other taxonomic groups. We suggest that the monitoring of the effects of forest management on biodiversity should use a "community composition approach" rather than the number of species (or genera or families) *per se*.

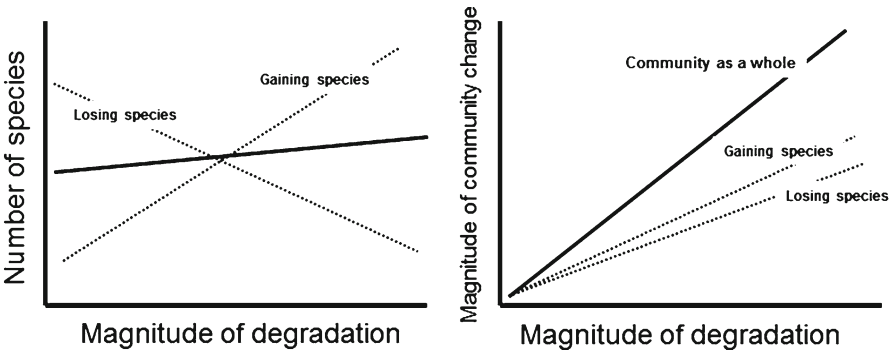


Fig. 8.1 Technical considerations in biodiversity monitoring. When biodiversity is monitored as the number of species (taxa) against the magnitude of forest degradation (as in the *left diagram*), it may result in an erroneous evaluation of a net increment because of a greater number of "newcomer gaining species" than "losing species." Effects of forest degradation can be correctly evaluated with a community approach where both gaining and losing species are taken into account (as in the *right diagram*)

The community composition approach is explained in Fig. 8.1. A community composition is the product of species assemblage (i.e., a list of species in a given area) and the abundance of each species (such as the number of individuals or the biomass of each species in the area). The overall change brought about by a management scheme is the sum of the magnitude of the reduction in loss of species and of the magnitude of increment in gain of species, which equals the change in species composition. The magnitude of reduction or increment is expressed by the change in species abundance. Technically, a change in community composition can be analyzed with one of several multivariate analyses. Community composition is sensitive to the magnitude of logging for all taxa that we studied in Chaps. 3–5. "Species" can be substituted with genus or family in any taxonomic groups. We suggest that the monitoring, reporting, and verification of biodiversity changes in future forest-related projects be based on the community composition approach.

8.4 Implications for Production Forestry in Borneo and Elsewhere

Borneo is the world's third largest island, with an area of 757,100 km^2, and it is divided administratively between Indonesia (states of East, South, Central, and West Kalimantans), Malaysia (states of Sabah and Sarawak), and Brunei. In recent years, Borneo has lost a large area of tropical rain forest because of forest fire, land conversion, and commercial logging (Langner et al. 2007). There are no reliable statistical data on the extent of intact forests left in Borneo. Using satellite data, the latest estimate of forested area, including regions of moderately degraded forests, ranges

from 40% (Darmawan 2004) to 57% (Langner et al. 2007) of the total land area. The majority of the remaining forests comprises production forests that have undergone commercial selective logging more than twice. Indonesia and Malaysia together (including their regions other than Borneo) accounted for about 42.8% of the CO_2 emitted from all land-use change and forestry (LUCF) activities in 2000 (WRI 2007). Because Borneo is the center of current world LUCF activities, the role of Borneo in the world carbon budget cannot be overstated from these figures.

Forest set aside in Borneo for the commercial production of timber resources occupies 37,500,000 ha (excluding Brunei), or 50.2% of the land area. The major proportion of this production forest is believed to consist of moderately or highly degraded forest stands or completely deforested stands because of past logging episodes of various intensities as well as forest fires. Many of these forests are legally protected for the future for permanent timber production. On the other hand, strictly protected areas do not exceed 10% of the land even if Brunei (a green state) is included.

Many faunal species, particularly large wildlife with wide home ranges, rely on degraded production forests in the current landscape of Borneo. In the case of the orangutan, nearly 70% of the wild population in the northeastern state of Sabah is found in production forests (see Chap. 6; Ancrenaz et al. 2005), and a similar trend exists for many endangered mammals (see Chap. 5; Venter et al. 2009). Recent studies in Borneo demonstrated that more than 90% of species recorded in primary forest were also present in logged-over forests (Berry et al. 2010) and more than 75% species of avifauna and dung beetles persisted in twice-logged forests (Edwards et al. 2011). Production forests are de facto reservoirs of biodiversity on the degraded landscape of Borneo. However, increased logging intensity will certainly cause negative consequences for many taxonomic groups, as we have demonstrated in Chaps. 3–6. The sustainability of production forests can thus influence the future carbon budget as well as the fate of wildlife in a global context.

In this volume, we demonstrated that improved forest management added to the baseline scenario of conventional logging has brought about a net carbon increment of 12 Mg C/ha per year (see also Putz et al. 2008) as well as the recovery of biological communities for many taxonomic groups. Our findings potentially apply to 37,500,000-ha production forests in Borneo and elsewhere in the tropics. If we straightforwardly apply the net increment of 12 Mg C/ha per year to 37,500,000-ha production forests, the total carbon additionality (improved management minus baseline management) will be 0.45 G ton C per year for all of Borneo.

8.5 Some Economic Considerations

The opportunity cost of sustainable forest management is the cost of improving forest management using RIL guidelines in comparison to conventional logging. The cost of implementing RIL guidelines was $135.00/ha in the case of Sabah (Pinard et al. 2000). Reduced revenue because of reduced timber yield may further add to the cost. Improved forest management has indeed not been widely adopted

elsewhere because of reduced net revenue (Putz et al. 2000). Currently, certified logs in Deramakot fetch a premium price and are sold for a higher price (Kollert and Lagan 2005). However, it is a rather rare case.

Sustainable forest management is the only pragmatic way to maintain the biomass, productivity, and other ecosystem functions of legally designated production forests. Reducing Emissions from Deforestation and Degradation Plus (REDD+) is receiving considerable attention as a post-2012 Kyoto mechanism to compensate developing countries for reducing CO_2 emissions from deforestation and forest degradation (Ebeling and Yasué 2008). If the carbon additionality in production forests (i.e., an increment of carbon over a baseline in a project period) is incorporated into REDD+, it will provide foresters with a maximal incentive to adopt improved forest management practices and will result in a higher permanent stock of carbon and better biodiversity conservation while maintaining a certain level of timber supply. If the economic value of carbon can be proportionately weighted by a net increase of biodiversity relative to a baseline, the total economic value would increase further. For this purpose, a valid biodiversity monitoring, reporting, and verification system needs to be developed.

8.6 Conclusions

1. Forests designated especially for the purpose of timber production exceed 50% of the land area in Borneo. Sustainable forestry of these Bornean production forests has a considerable influence on world LUCF carbon emission, as well as the fate of biodiversity, especially endangered wildlife.
2. Sustainable forest management that employs best management practices, such as long-term management planning, RIL, and strict compliance with international principles, can maintain the carbon stock at a much higher level than does baseline conventional logging. Sustainable forest management can also restore the integrity of communities of various taxonomic groups and thus has favorable co-benefits for biodiversity conservation.
3. Monitoring of biodiversity in a few representative sites is essential to secure the co-benefits of sustainable forest management. The magnitude of a change in community composition can best indicate the complex effects of deforestation and forest degradation on biodiversity.
4. It is highly desirable to include the monitoring of biological communities based on community composition as a means of monitoring biodiversity in forest certification schemes (e.g., Forest Stewardship Council). The inclusion of such a biodiversity metrics will discourage the perverse incentive for unsustainable production forestry, which can be otherwise falsely certified.
5. Practicing sustainable forestry with strict compliance to internationally accepted principles and standards is the best mitigation option to reduce the current rate of carbon emission and biodiversity deterioration. This approach seems to be the most sensible approach where production forests dominate the landscape. If the concept of additionality (i.e., an increment of carbon over a baseline in a project

period) is incorporated into REDD+, it will provide foresters with a maximum incentive to adopt sustainable forest management and will result in a higher permanent stock of carbon and better biodiversity conservation.

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